**Rocky Flats Environmental Technology Site**
**Site-Wide Water Balance Project**
**Code Validation Program**

# MIKE SHE PAPERS AND PUBLICATIONS
## Volume 1
## 1994-1996

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# *MIKE SHE Papers and Publications – 1999-1994*

## DHI 1999


Styczen, M., Thorsen, M., Refsgaard, A., Christiansen, J.S. and Hansen, S. (1999) Non-point pollution modelling at different scales and resolution, based on MIKE SHE. *Proceedings of DHI user conference 1999.* DHI ref. 27/99.

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DHI Software Conference, 1999

# Non-point pollution modelling at different scales and resolution, based on MIKE SHE

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## Abstract

The spread of tools for assessing the contribution of non-point pollution to groundwater or surface water systems range from very simple to very advanced models. The simple models do not, in general, take into account the annual variations in weather and they are seldom able to incorporate effects of management in a realistic manner. The advanced models require extensive parameterisation. Furthermore, important decisions are required regarding the size of area, the resolution (grid size) used in the model, and the level of details required for the input. The article provides an overview of strategies used at the Danish Hydraulic Institute in non-point pollution modelling when different scales are considered, moving from small study areas to small catchments to regional scale or vice versa. These strategies were applied in former and ongoing projects. Furthermore, effects of different resolutions on a given catchment are exemplified. The influence of the level of detail in the data source can be evaluated from two studies carried out.

In general, scaling up with respect to size of area (of similar type) can be done with reasonable results for nitrate. Simulations of a particular catchment with a given set of base data, but with different grid resolutions, demonstrates that the simulated river discharge hydrograph strongly depends on the grid resolution, whereas annual discharge values and aquifer nitrate concentrations are less dependent on grid resolution. Regarding data resolution, the experience with regional databases is not unidirectional. For areas with little variation in landscape factors, regional simulations of nitrate concentrations in groundwater were adequate. For more complex areas, particularly due to a more complex geology, the information was inadequate.

## 1 Introduction

DHI's experience in non-point pollution modelling mainly relates to nitrate and pesticides. The involvement in non-point pollution modelling began in 1990, with modelling of nitrate at catchment scale under the Danish "Nitrate, Phosphorus, and organic matter" research programme. The programme was a consequence of the increasing eutrophication observed in the coastal zone of Denmark, but also in lakes, streams and to some extent in groundwater.

The work on nitrate modelling was (and is) done in close co-operation with the Royal Veterinary and Agricultural University in Copenhagen, who developed the DAISY-model for nitrate modelling. The experience gained was later used in other projects in Denmark and

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Slovakia, and DHI is presently involved in nitrate modelling in Poland and at another Danish site. A common feature of the projects has been calibration of model components on plot scale or at least on small catchments along with upscaling to larger areas.

The major strength of the advanced N-modelling in comparison to simple estimates is the possibility of evaluating effects of farm management changes on nitrate leaching. In addition, comparison between simulations and measured data allows distinguishing between effects of management and effects of weather.

The work on modelling of transport and transformation of pesticides started a few years later, and has, until recently, concentrated on simulation of single columns (point scale). However, as the ongoing national monitoring programme documents presence of pesticides in groundwater and surface water, the need for modelling of pesticide dynamics in at least small catchments has grown.

## 2 Models

The non-point pollution modelling conducted by DHI has involved the use of the hydrological model system MIKE SHE (Abbott et al., 1986 a,b; and Refsgaard and Storm, 1995). MIKE SHE functions as a catchment-modelling tool, able to simulate flow and solute transport. Different modules describing the reactions of the solutes are then added to this description.

For pesticides, these process descriptions have been implemented as part of the MIKE SHE system from the beginning. For nitrate it has been different: In the beginning, the DAISY model, developed at the Danish Veterinary and Agricultural University, so closely produced the output of the unsaturated zone, subsequently transferred as input to the MIKE SHE system. Recently, the two models were integrated, so the reactions and temperature calculations take place in DAISY while the flow and solute transport take place in MIKE SHE. With respect to pesticides, another recent development is a further merging between the DAISY process descriptions and the pesticide processes: A module for microbiological degradation and sorption in MIKE SHE utilises the organic matter turn-over of DAISY to regulate pesticide turn over, also determined by redox conditions.

It is a causcious decision to link MIKE SHE and the DAISY model. DAISY has proven its quality in several model inter-comparisons, and its performance under Danish conditions is well documented. It is being maintained in its stand-alone form by the University, and is thus continuously updated, as new results become available. Essentially, the aim is to work with few model codes with a high degree of flexibility, in order to be able to cover as many practical situations as possible.

### *2.1 MIKE SHE*

MIKE SHE is at present the only physically based, dynamic, fully distributed modelling tool for integrated simulation of all major hydrological processes occurring in the land phase of the hydrological cycle. The combination of a physically based and a distributed model enables a direct use of field data for model building and it enables linking to spatial data that may, for instance, be provided through remote sensing or fields survey programmes. The integrated approach makes MIKE SHE suitable for simulation of hydrologic systems where surface water and groundwater interactions are significant.

The basic MIKE SHE module is the Water Movement module describing the hydrological processes. The hydrological components included are interception-evapotranspiration, infiltration, snow melt, 1-dimensional flow in the unsaturated zone, 3-dimensional ground water flow, overland flow in 2-dimensions and 1-dimensional river flow, all of which are fully coupled.

MIKE SHE can be combined with a variety of add-on modules used to address specific environmental problems among these solute transport by advection and dispersion (MIKE SHE AD). The integrated approach also covered by the solute transport and the module flexibility make MIKE SHE suitable for a large variety of environmental and hydrological issues. MIKE SHE is applicable on spatial scales ranging from a single soil profile to large regions, which may include several river catchments.

## 2.2 DAISY

DAISY is an advanced soil-plant-atmosphere system column model. It describes crop production as well as water and nutrient dynamics in the root zone of the agro-ecosystems according to various management strategies, including crop rotations, fertilisation, irrigation, soil tillage and crop residue management. The model simulates processes including: plant growth and crop production; heat flux and soil temperature; soil water uptake by plants and evapotranspiration; carbon and nitrogen mineralisation; nitrification and denitrification (nitrogen transformation); nitrogen uptake by plants.

Combined with the MIKE SHE WM and AD modules (*Figure 1*), the DAISY module provides a powerful tool for the assessment of the regional impacts of agricultural crop production system management on water quality conditions in the soil, the groundwater, and streams.

The DAISY module requires temporal information on air temperature and global radiation and information on humus and inorganic nitrogen content in the soil. As agricultural driving variables, information on crop rotations, tillage operations, fertilisation, and irrigation need to be specified at present. The DAISY module provides crop parameters for 12 crops, which are calibrated using data from crop varieties grown in northern Europe. To ensure that the crop development is simulated correctly, it may be necessary to recalibrate these parameters using local data on crop development.

The DAISY model is developed by the Royal Danish Veterinary and Agricultural University. Further development of the MIKE SHE DAISY is ongoing to include atmospheric processes and remote sensing. Additionally, phosphorus transformations and transport is expected to be included within a few years.

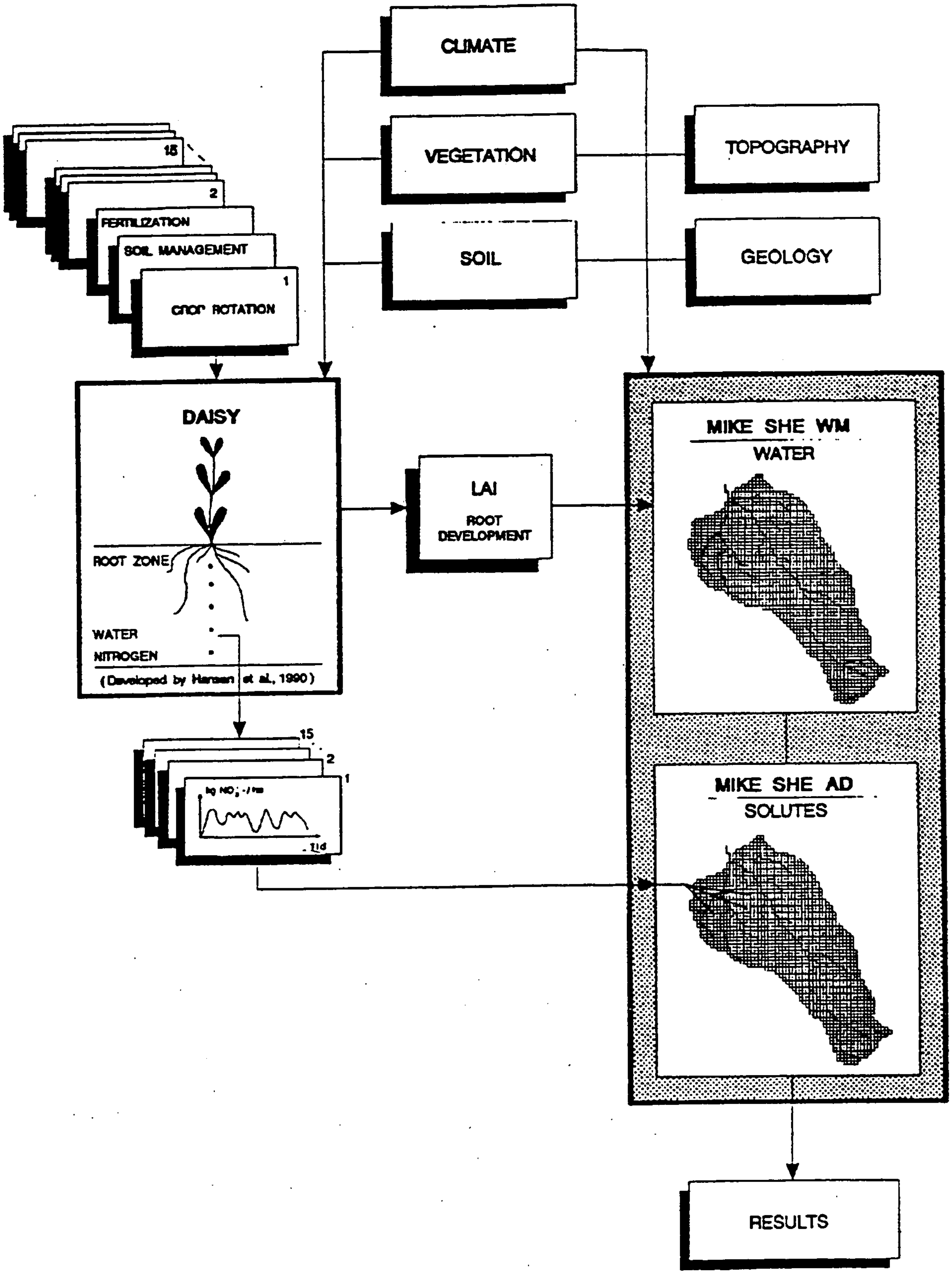


*Figure 1 The integration between DAISY and MIKE SHE. DAISY simulates evaporation, infiltration, temperature, growth, and nitrogen transformations. MIKE SHE simulates transport of water and solutes.*

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### 2.3 Pesticide

Modelling the transport and metabolism of pesticides needs incorporation of sorption and degradation processes in the transport process. Besides the physical transport by advection and dispersion these two chemical and microbiological processes are the most influential regarding the transport and fate of pesticides.

The MIKE SHE sorption/degradation (SD) module includes simplified descriptions of complex geochemical and microbiological processes. In combination with the MIKE SHE AD module it constitutes a model tool for describing the influence on pesticide transport by chemical and microbiological processes. Both linear and non-linear equilibrium or kinetic sorption are options dependent on reaction rates and distribution coefficients for the pesticides. A first order process describes the microbiological degradation of the pesticides with a certain half-life time constant. This description of pesticide metabolism has been used earlier in projects and might be sufficient with respect to most problems.

The MIKE SHE geochemical module (GM) and biodegradation module (BM) includes advanced descriptions of complex geochemical and microbiological processes, respectively. The GM module handles all inorganic equilibrium chemistry (precipitation, dissolution, complexation, ion exchange, redox, and sorption) by invoking the PHREEQC code (Parkhurst, 1995). The BM module handles various biological processes (0. order, 1. order, Monod kinetics) and can be tuned to handle solute specific expressions derived from laboratory experiments. In combination with the MIKE SHE AD module these two modules constitute an advanced model tool for describing the influence on pesticide transport by chemical and microbiological processes. This model system is employed in the most recent studies of pesticides that require the most careful and accurate description of the metabolism, e.g. a study on pesticides and groundwater, Section 4.2.2.

## 3 Approach to Non-point Modelling

Large-scale hydrological models are required for a variety of applications in hydrological, environmental and land surface-atmosphere studies, both for research and for day to day water resources management purposes. The complex interaction between spatial scale and spatial variability is widely perceived as a substantial obstacle to progress in this respect (Blöschl and Sivapalan, 1995; and many others). The process of upscaling does pose problems, particularly if it is attempted to describe the processes differently, depending on the scale of application. The method, which has been applied at DHI, does not employ different process descriptions, but rather an aggregation procedure. In the aggregation to macro-scale, the variations in spatial data such as soil types and crops are preserved statistically, although they are not correctly georeferenced.

In all applications, the same modelling system is used whether the simulation represents field or plot scale, a small catchment or a region. However, the models differ with respect to area covered, resolution (grid size) and the level of detail available with respect to input data. All three factors influence the result of the simulation.

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It is of outmost importance that the model code has been validated at plot scale, preferably under different conditions. This way it is ensured that the process descriptions are appropriate and that realistic results can be obtained with a physically relevant parameterisation. Figure 2 illustrates some of the complexities of non-point pollution pathways and scales. Plot scale means, in this sense, not necessarily point scale, but rather a field scale characterised by "effective" soil and vegetation parameters, but assuming only one soil type and one cropping pattern. Thus, the spatial variability within a typical field is aggregated and accounted for in the "effective" parameter values (Refsgaard et al., 1998). MIKE SHE and DAISY have documented their ability to describe conditions at field scale (e.g. Jensen and Refsgaard, 1991

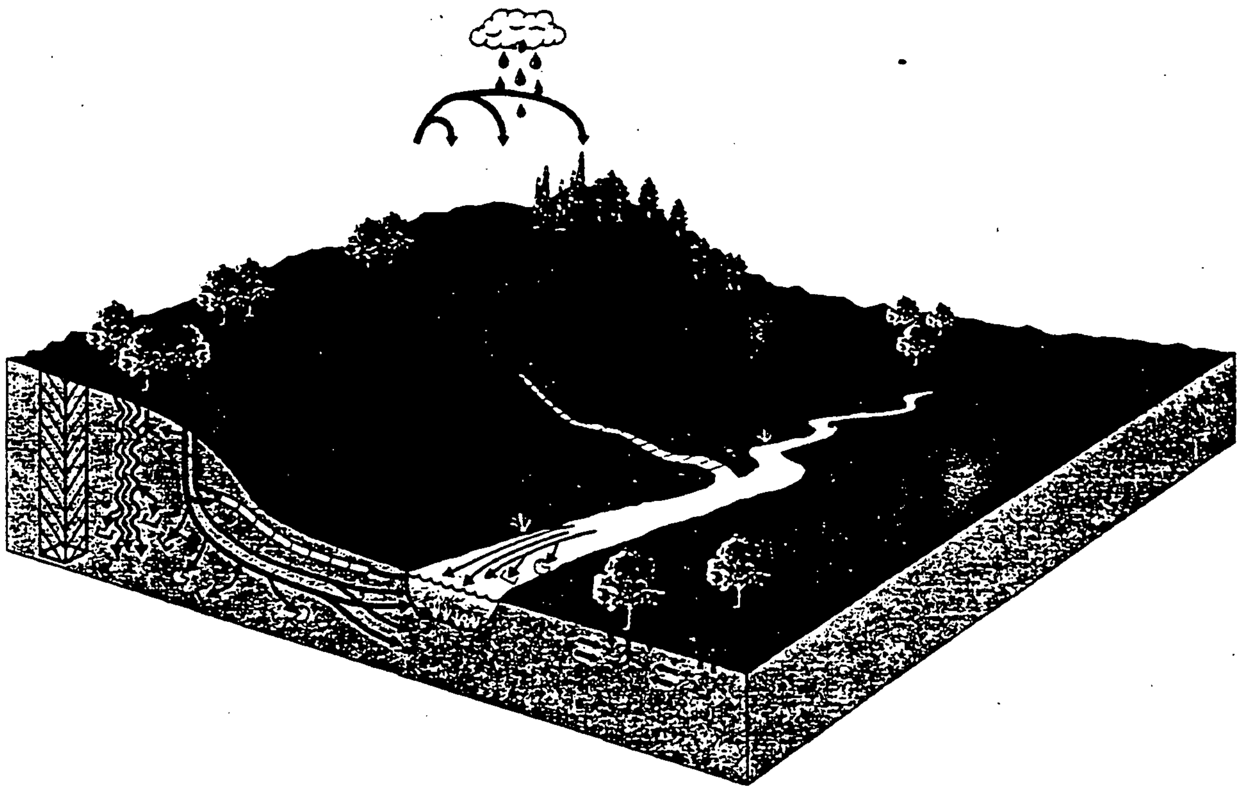

*Figure 2. Simulations of processes in a catchment may take place at plot, local or regional scale, symbolised by a soil column, a subcatchment or the whole area. Relevant processes for non-point pollution may include transport along the soil surface, through the soil matrix or macropores, in drains and in groundwater, of water (blue) and solute in soluble (red) or particle bound (brown) form, as well as several types of transformations. Transformations e.g. be degradation, plant uptake, or fixation through sorption or immobilisation. Some chemicals are subject to drift and volatilisation.*

a, b, c; Djurhuus et al., 1999; Svendsen et al., 1995, Diekkrüger et al., 1995; and Willigen, 1991).

For field or plot scale simulations measured data are employed to the largest possible extent. Calibrations are carried out on available data, such as soil moisture measurements, drain flow, or solute concentrations. It is a rule at all scales of simulations that the flow-relevant parameters are calibrated first (groundwater levels, drain flow, river flow, moisture in the soil profile), followed by calibration of conservative tracer, if data are available, and finally the chemical in question. A strong advantage of utilising one model system is that the implemented model can be tested on detailed data when these are available.

When moving to a small catchment, measured data are still utilised to the greatest possible extent, but it becomes necessary to generalise information into a number of types (soil, crop rotations, fertilisation times, etc.). The level of information available for a column is usually less than at the detailed level. The grid size is usually larger.

At the regional scale, the model parameters are based on best guesses constructed from statistical information, interviews, national databases, etc. Parameters calibrated from detailed studies will form the basis for qualified guessing, but the availability of precise information is (considerably) less than at the plot scale and generally less than in the small catchment. In principle, the grid resolution could be the same as for the small catchment. In practise, however, it will usually be larger. The grid resolution issue is discussed in Section 4.3.

The use of statistical data is a key issue in the data handling at the regional scale. It is employed in two ways:

- The statistical information is utilised for generation of a number of realistic crop rotations, producing the crop cover actually observed, and utilising the fertilisers consumed and the manure produced within the catchment. The amount of land under different rotations has to fit the distribution of farm types, taking into account that some types of rotations are typical for a cattle farm, others for mixed farms or for pure plant growing farms. Furthermore, the fertilisation schemes were derived for each crop based on whether the farm type produced manure, and what amounts were usually applied to the crop in question. The statistical information is thus disaggregated to produce an "intelligent" guess of the most likely distribution of factors in the study area.
- Certain parameters are highly variable, and to obtain an estimate of the effect of their variations, it is possible to conduct sensitivity analysis or Monte Carlo simulations. A sensitivity analysis provides an estimate of the uncertainties caused by a single parameter, while a Monte Carlo simulation provides an estimate of the uncertainty provided by random variations in different parameters at the same time. A sensitivity analysis can be used for pinpointing the most important parameters to be included in a Monte Carlo simulation The simulation results become an interval or a band rather than a single value. At the regional scale Monte Carlo simulations are exemplified in Section 4.4.

While upscaling receives attention in literature, the process of going from a coarser grid system in a regional model to a fine grid system in a local model is seldom considered. In practice, a local model, perhaps with very detailed local information regarding agricultural input, suffers from strong effects of boundary conditions. One approach to handle this is to let a regional model provide the boundary conditions to the local model. While this approach is considered superior to simple estimates, it has not yet been possible to quantify the derived uncertainties with respect to solute transport. However, the approach has been successfully used in hydrological studies (Refsgaard et al., 1994a and b).

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# 4 Experiences of Non-point Pollution Simulations at Different Scales.

## *4.1 Performance at Point Scale*

The first issue to consider is the validity of the model on the plot scale. This is important, not least to document that the process descriptions included in the model code are adequate. However, it is not always possible to obtain a full set of local data at plot scale and one may have to rely on a rougher calibration and results of preceding model validation exercises. One example of a DAISY validation exercise is shown in *Figure 3*.

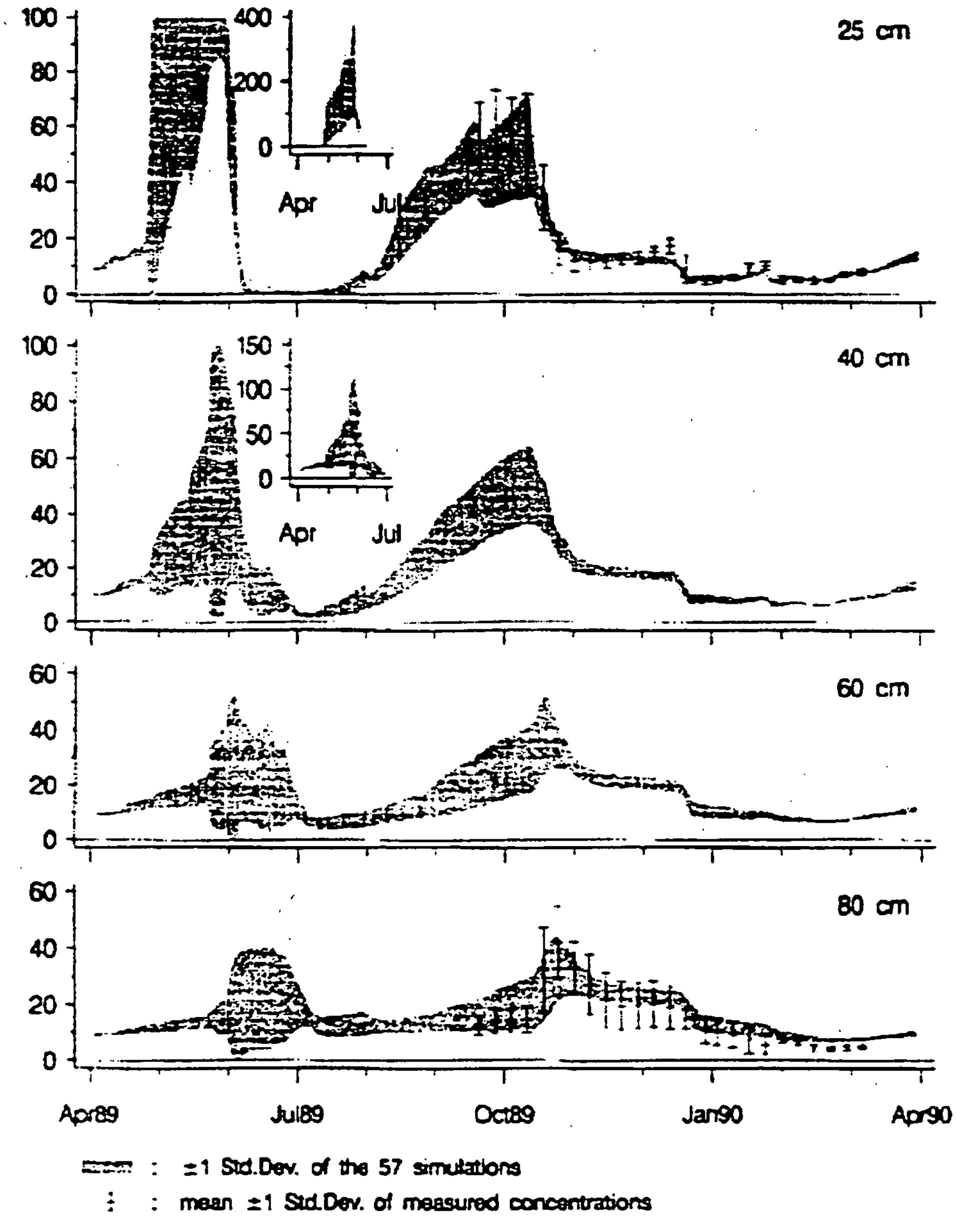


*Figure 3. Simulated and measured concentrations of NO3-N (mg/l) (y-axis) at 57 points on a Jyndevad soil. Djurhuus et al. (1999) p. 272. Copyright (1999), with permission from Elsevier Science.*

The figure shows the distribution of nitrate concentration measured for 57 points, compared with the distribution of 57 simulations based on locally measured parameters from the same 57 points in the field. Thus, two distributions are compared, rather than only the variation on model results compared to one set of field measurements. The correspondence between the two is impressive. As mentioned in Section 3, a substantial amount of literature documents DAISY's performance.

For pesticides, the adequacy of the simulations is less well established. DHI has been involved in a few studies concerned with evaluation of this topic for Danish conditions (e.g. Styczen and Villholth (1994). The most recent study is an evaluation of MIKE SHE SD performance for simulating one-dimensional pesticide leaching from lysimeters (Jørgensen et al., 1998).

Within the most recent project, the performance of MIKE SHE SD was compared to three widely used pesticide leaching models, PELMO (Klein, 1995), PESTLA (Boesten, 1993) and MACRO (Jarvis, 1994). Simulation of lysimeter data was conducted in two steps. First, a blind simulation without any calibration allowed was carried out and second, water and conservative solute transport processes were calibrated using measured outflow of water and bromide. No calibration of laboratory determined pesticide parameters in terms of linear sorption coefficients ($K_d$) and degradation half-life times ($T_{\frac{1}{2}}$) was allowed during these steps. Simulation results were evaluated using statistical tests and the outcome of these were compared to pre-determined performance criteria suggested by a European workgroup on regulatory use of pesticide leaching models (FOCUS, 1995). Further details on data background, statistical test procedures, model development, and model performance are available in Jørgensen et al. 1998 and Thorsen et al. 1998. Key results from the statistical evaluation are shown in *Table 1*.

*Table 1. Results of the model performance test for simulation of Mecoprop leaching from lysimeters. The blind simulations show model performance using measured parameters without any calibration. The calibrated simulations show model performance after calibration of water and Bromide transport. No calibration on pesticide parameters was allowed in these two steps. Shadings indicate the step, if any, in which the selected performance criterion was met. A "÷" indicates that the factor-of-F test was not successful in any step.*

| Data type: | | Accumulated percolation | Accumulated MCPP leaching | MCPP peak concentration |
|---|---|---|---|---|
| Statistical test: | | % deviation | Factor-of-F test[1] | Factor-of-F test[1] |
| Performance criteria: | | 8 % | F = 2 | F = 2 |
| PELMO | Blind simulation | 10 | ÷ | ÷ |
| | Calibrated | 10 | ÷ | ÷ |
| PESTLA | Blind simulation | 13 | ÷ | ÷ |
| | Calibrated | 6 | ÷ | ÷ |
| MACRO | Blind simulation | 8 | F = 2 | F = 2 |
| | Calibrated | 2 | F = 2 | F = 2 |
| MIKE SHE | Blind simulation | 12 | F = 2 | F = 2 |
| | Calibrated | 7 | F = 5 | F = 2 |

[1]The factor-of-F test compare simulated values with the range of measured values from the 2 lysimeters. F = 2 or F = 5 indicate that simulation results lie within a factor of 2 or 5, respectively, from the measured values. The statistical tests are further described in Jørgensen et al. (1998) and Thorsen et al. (1998). MCPP: Mecoprop.

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The experimental data showed that water flow and pesticide leaching was primarily controlled by preferential flow through a small active pore volume. The main results of the model performance test was that the two model codes containing a description of preferential flow (MACRO and MIKE SHE) passed the performance criteria for pesticide flux already in the blind simulation whereas PELMO and PESTLA required calibration which violated the original parameterisation (results not shown).

## *4.2 Modelling at Local and Catchment Scale*

Several studies contain the approach of moving from plot scale to small catchment or regional scale. *Table* 2 contains a sample of studies conducted with DHI involvement showing the area simulated, the resolution used, and the data sources.

*Table 2. Overview of scale, resolution and data sources used in different studies presented.*

| | Scale | Grid size | Data |
|---|---|---|---|
| **NPo-project-Karup** | | | |
| Experimental plots | plot size | plot size | Local |
| Rabis Creek | 16 km$^2$ | 400 m | Local and Regional/ statistical |
| Karup Stream catchment | 440 km$^2$ (groundwater catchment) | 500 m | Regional/ statistical |
| **Pesticides in groundwater** | | | |
| Karup Stream catchment | 440 km$^2$ (groundwater catchment) | 1 km | Regional/statistical |
| Fladerne Creek | 4 km$^2$ | 100 m | Local |
| Vaarby Stream | 360 km$^2$ | 1 km | Regional/statistical |
| Lungrende Creek | 4 km$^2$ | 100 m | Local |
| **Pesticide in surface water** | | | |
| Intensively monitored fields | plot size | plot size | Local |
| Lillebæk catchment | 4.7 km$^2$ | Not yet determined, approx. 25 m | Data for each field. Some data will be treated statistically. |
| **Brøns creek** | | | |
| local scale | 10 km$^2$ | Not yet determined, approx. 50 m | Local |
| regional scale | 100 km$^2$ | Not yet determined, approx. 200 m | Regional/statistical |
| **UNCERSDSS*** | | | |
| Karup Stream catchment | 518 km$^2$ (topographic catchment) | 1, 2, 4 km | EU-scale |
| Odense Stream catchment | 536 km$^2$ | 1, 2, 4 km | EU-scale |

*UNCERSDSS: Uncertainty of spatial decision support systems, EU reseach project.

### 4.2.1 The Karup study

The first study, which was conducted with MIKE SHE/DAISY. took place in the Karup catchment in Jutland (Storm et al., 1990: Styczen and Storm 1993a and b). In practice. simulations took place at three scales, as shown in *Table 2*. The physical location of the Karup catchment is shown in *Figure 4*.

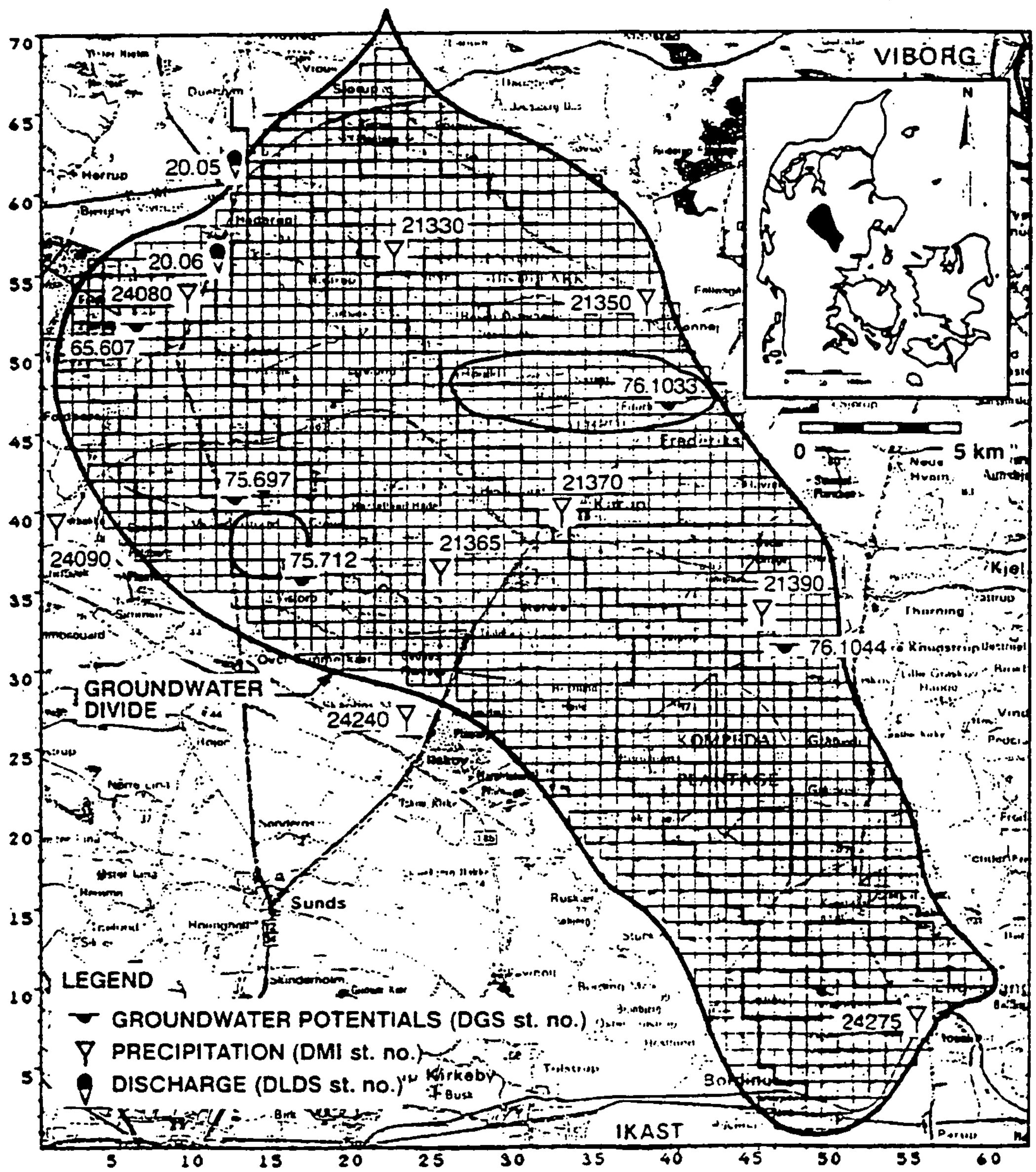


*Figure 4 The Karup groundwater catchment. The catchments for Rabis Creek and Fladerne Creek, also mentioned in the text, are indicated with red, Rabis creek towards the east and Fladerne creek to the west.*



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The plot scale simulations were carried out by DHI as part of the initial work of setting up the regional model, but were never published. For Rabis Creek, detailed simulations of water flow, and tritium movement were carried out, together with some geochemical modelling of nitrate, mainly concerned with processes taking place along the redox front by Engesgaard and Jensen (1990).

The regional model is based on the results from the plot simulations and the detailed study of Rabis Creek. Furthermore, it is based on data from local meteorological stations, all available map information (soils, topography, land use), statistics on fertiliser use, number of animals, and farm types, interviews with agricultural consultants, interpreted geological profiles and geological data from boreholes, stream flow data, and groundwater head, monitored during part of the simulation period.

The data interpretation included, among others, positioning of permanent grass areas in areas with shallow groundwater identified through preliminary simulations with MIKE SHE. The agricultural data were interpreted as described in Section 3. Similarly, the tillage methods chosen and the methods of manure storage assumed changed over the simulation period according to the information given and the crop simulated. Land use and the distribution of crop rotations are shown in *Figure 5*. The redox front was interpolated from information from the 120 boreholes.

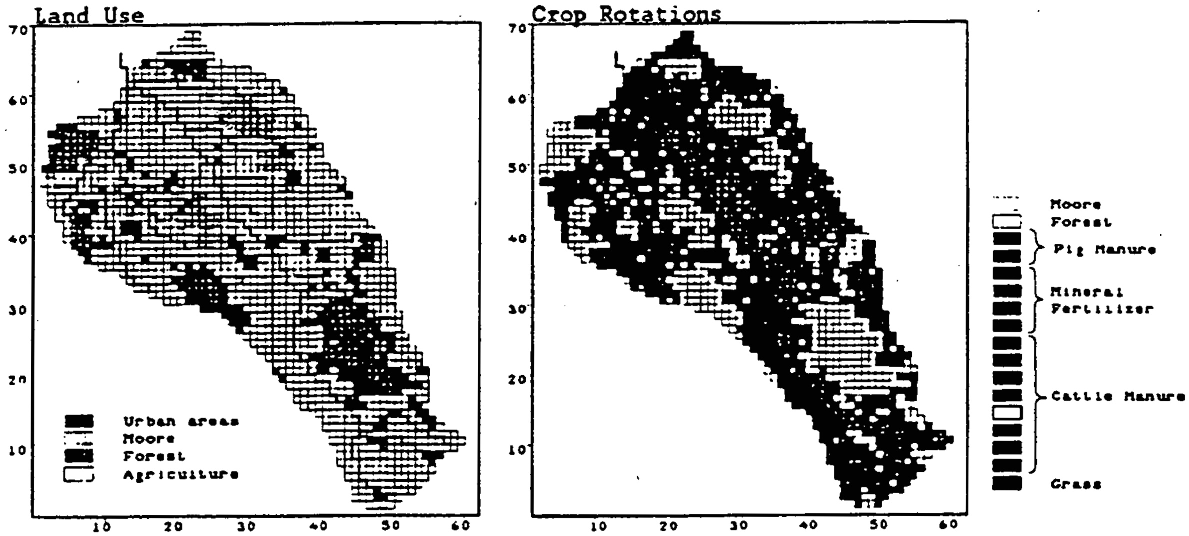


*Figure 5* *Distribution of crop rotations in the Karup catchment.*

To test the adequacy of the regional model, the Rabis Creek subcatchment was simulated with smaller grids, and the results were compared to actual measurements in the groundwater. As the data in the regional model was based on statistics, the results could not be expected to be correct in a given point. Two simulations were carried out: a) with the statistical input, and b) with the average of the statistical input for all cells. The data sources and in the latter case, the treatment are identical for Rabis Creek and the Karup catchment and the grid resolution is almost identical. The simulated result and the observations showed a large degree of accordance, when the discretisation of the saturated zone was taken into account. Features such as effects of rather wet or dry years clearly showed up in the results. The distribution of nitrate in time and space is documented in Storm et al. (1990) and Styczen and Storm

  

(1993b). With good results in Rabis Creek, it was expected that the regional simulations would be adequate, as the grid size was almost unchanged, and the database was identical. Only the size of the area differed. An example of the output is shown in *Figure 6*.

While the overall flow parameters of Karup were simulated very well for the catchment the simulated nitrate concentrations in the river were much larger than the observed. The plausible reason for this is the high denitrification, which takes place in the wetland areas near Karup Stream. This process was not included in the model at the time.

A similar analysis was later carried out for Danube Island in Slovakia (Refsgaard et al., 1998).

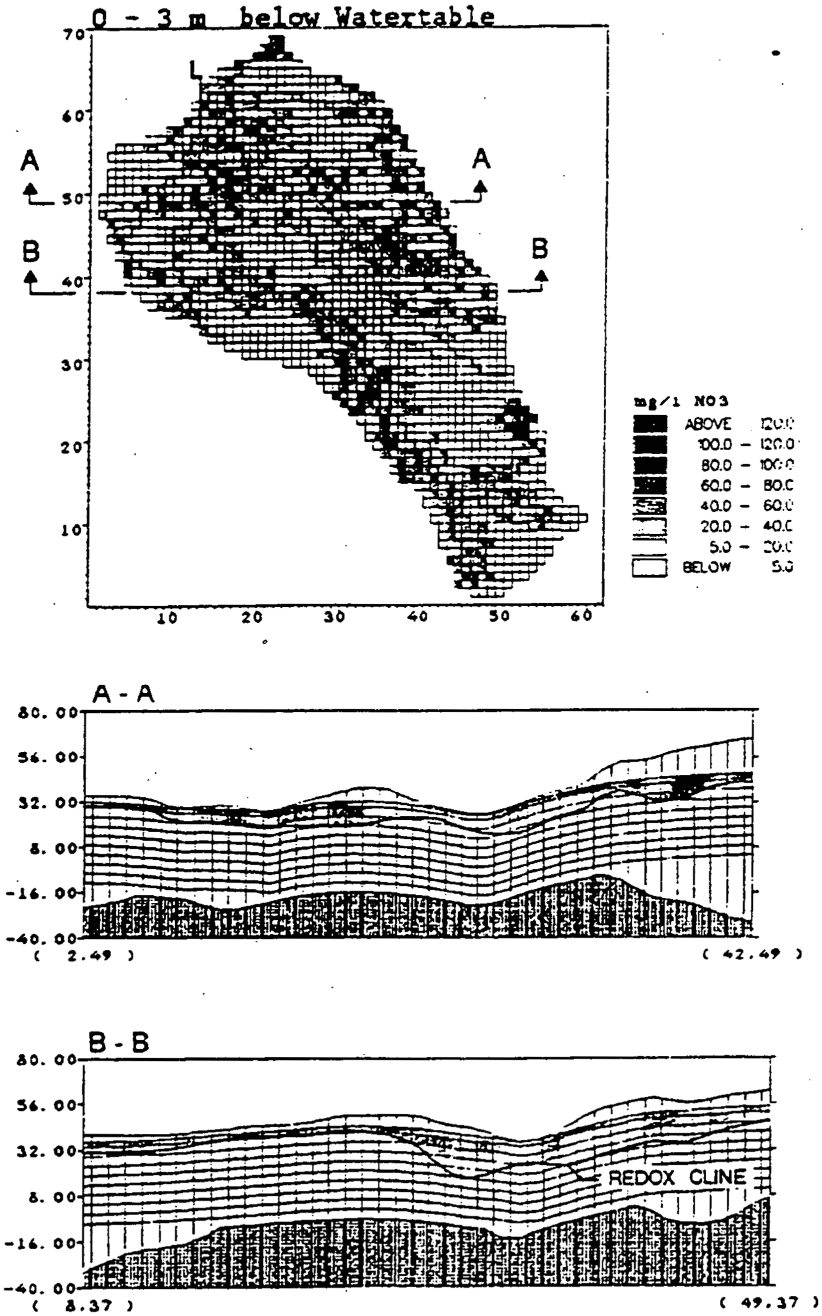


*Figure 6. An example of distribution of nitrate concentrations in the groundwater of Karup catchment.*

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### 4.2.2 Pesticides and groundwater

In an ongoing project under the strategic environmental scientific program by the Danish Ministry of the Environment the scaling problem is being investigated within the frame of agricultural use of pesticides and the impact of the groundwater. Three target pesticides are in focus: Mecoprop, Bentazon, and Isoproturon. Field investigation and subsequent modelling is taking place on two different scales. In this case the scaling problem refers to the differences in the area extent and data resolution.

Two field sites have been chosen as representative for Danish conditions. The Karup catchment in the central part of Jutland and the Vaarby catchment in the western part of Zealand have been selected as representatives for large scale sandy and clayey till (overlain by sandy loam) soils, respectively. Within these two catchments two sub catchments (the Fladerne catchment in Jutland and the Lungrende catchment at Zealand) have been selected to reflect the small catchment scale. The area extent of the catchments is shown in *Table 2* and the location of Fladerne Creek in *Figure 4*.

The pesticide input function and the validation of the models depend on the scale. In the case of the two small catchments, interviews have been conducted with the local farmers and their consultants to map the use of crop specific pesticides during the last few years in detail. Also, monitoring of the pesticide content in the groundwater and in the stream discharge is being performed to enable validation.

A similar procedure with interviews is not possible at large scale. Hence, the pesticide input function is created by the use of agricultural statistics of the general crop use and crop specific pesticides during the last decade in the area. A similar line of approach was used in the NPo-project (Section 4.2.1) with respect to the agricultural use of fertiliser. No monitoring of the pesticide content in the groundwater and in the stream discharge is being performed.

Preliminary results from laboratory investigations have indicated that microbial degradation takes place in the aerobic zone of the hydrological system. However, the target pesticides persist when oxygen is not present. Also, it is shown that the pesticide sorption among others depends on the concentration of pesticides. Laboratory experiments have indicated that the three target pesticides do not behave identically with respect to process description and parameterisation. Thus, the application of an advanced model for transport and metabolism seems to be of importance.

### 4.2.3 Pesticides in surface water

The objective of this project is to produce a tool for evaluation of pesticide in surface water, to be used by the Danish Environmental Protection Agency, in connection with pesticide registration.

In this project, which is still in its initial phase regarding modelling, pesticide will be simulated on plot scale and in a small catchment of 4.7 $km^2$. A detailed monitoring programme for 6 experimental plots, on which all input is known (pesticides, fertiliser, manure, tillage, etc) provides validation data. Moisture, drain flow, groundwater level, nitrate

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in soil moisture and nitrate and phosphorus in drain flow and groundwater are being observed frequently. Pesticide is also monitored in connection with some of the plots. The information from the plots will be utilised for calibration of water flow and nitrate-related processes (inclusive of solute transport). Phosphorus will be used as an indicator for colloid transport in macropores and drains, which also takes place with respect to pesticides.

For the small catchment, detailed information will be available concerning crops and spraying practices on all fields. Pesticides are monitored in drains and in the stream. In this case, the data related to agricultural input exists almost as detailed for the entire area as for the plots. However, the soil information available and of course the calibration/validation data are found on a larger scale. The resolution of the final setup is expected to be below field size (i.e. 25x25 m).

Regarding transport pathways, the study is most comprehensive, as drift, macropore flow, colloid transport and soil erosion are considered, on top of the usual pathways. This project may be followed on http://projects.dhi.dk/pesticide.

### 4.2.4 Brøns Creek – Generating boundary conditions to a local model

DHI has used regional hydrological models as boundary conditions for more detailed models for some time. Regarding solute transport, however, it is a new approach. In a recently started project in Brøns Creek the approach will be tried out.

A river restoration project for a small section of the Brøns creek in the Southern part of Jutland, Denmark, has been initiated. The section is about 5 km long with a topographical catchment area of approximately 10 $km^2$. It is the lower part of a hundred-$km^2$ river catchment mainly covered with farmland and plantation. The project goal is to increase the potential for $NO_3$ removal in the wetland areas along the creek.

Measures to increase the denitrification processes include restoration of the old creek course water level rise in the creek by "semi-natural" submerged weirs, and cutting off all artificial drainage pipes and ditches in the catchment. It is planned to apply the models mentioned earlier on different scales to demonstrate the possible effect of the restoration project.

On the 10 $km^2$ scale (local scale) it is possible to obtain detailed information on land-use for each field in the catchment for the last five years both with respect to crop type and fertiliser type and amount. This information forms the main input to MIKE SHE DAISY.

It has been decided to construct a model covering the entire one hundred-$km^2$ catchment (regional scale) for two reasons:

1. To generate realistic, time-varying boundary conditions to the local scale model in terms of water and $NO_3$ fluxes and groundwater head variations,
2. To allow calibration of the model on measured discharges and $NO_3$ fluxes in a gauging station located at the outlet of the local (and regional) scale area.

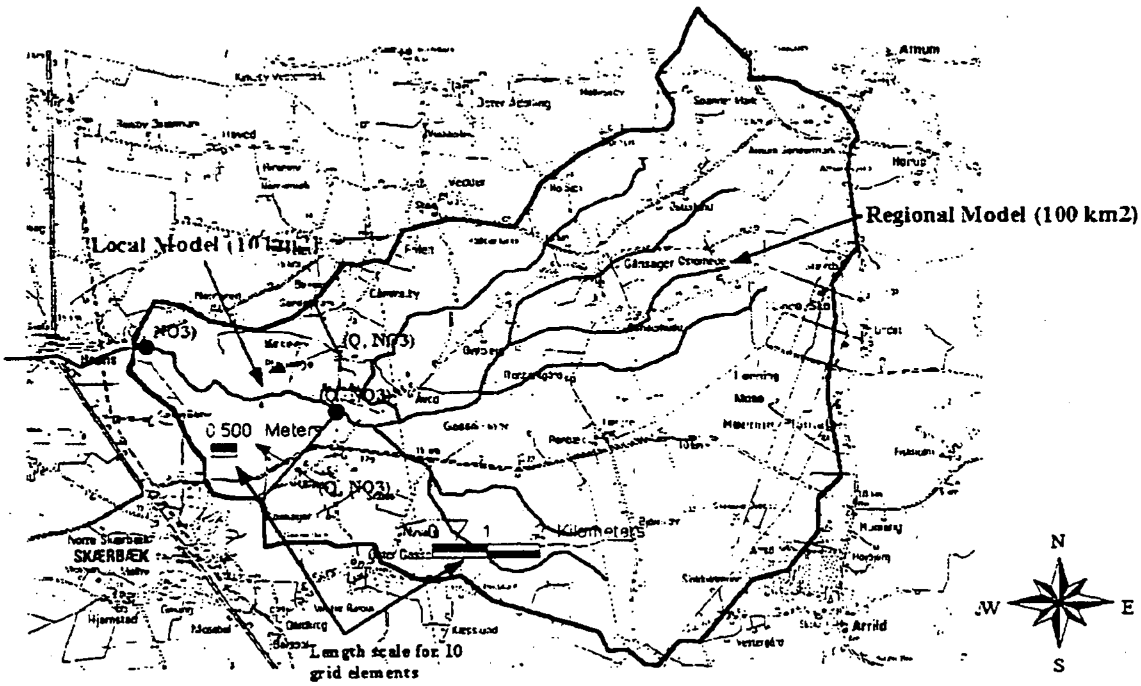


*Figure 7. The regional (black boundary) and local (green boundary) catchments for Brøns Creek. The resolution is tentatively assumed to be 50 and 200 m for the local and regional model, respectively. The two scales on the figure thus indicates the length of 10 grids for each of the two models.*

For the larger area it is impossible to obtain deterministic values of land-use and fertiliser application and the statistical approach was applied to allow DAISY to generate infiltration of water and $NO_3$ to MIKE SHE. Firstly, the area will be divided into main land use types – farmland, plantation, urban areas and areas with permanent grass. Secondly, the statistics and the agricultural information available will be used to define sets of crop rotations and cropping practices, which will be distributed randomly over the farmland, as explained in Section 3 and 4.2.1. A simple approach for $NO_3$ degradation in the groundwater zone was introduced similarly to the approach used in the Karup study (4.2.1).

The regional simulation will be used to generate the boundary conditions in the form of water and solute fluxes for local scale modelling.

### 4.3 Effects of Grid Resolution

The smallest horizontal discretization in a distributed model is the grid size, which is typically larger than the field scale at which parameters are determined. Running the model at large grid resolutions but using model parameters valid at field scale is necessary to make the computational demand acceptable for large catchments or regional scale applications. A critical question is, therefore, how the catchment scale model output is influenced by selection of grid resolution. The following comparisons concern grid sizes of 1, 2 and 4 $km^2$.

Two study areas were included in the analysis: the Karup stream catchment and the Odense stream catchment. The main differences between these two catchments are that where the Karup catchment is characterised by flat topography, coarse topsoil texture, and rather uncomplicated geology with unconfined conditions in the aquifer, the Odense catchment has a more varied topography, topsoils belonging to more fine textured classes, and more complex geology with confined groundwater conditions. The catchment characteristics, data background, and model construction are further described in UNCERSDSS (1998) and Refsgaard et al. (1999). An important difference to the earlier Karup study is that data, where possible, was obtained from European data bases which influenced the model construction. For example, the topographic catchment rather than the groundwater catchment was delineated and the geological description was considerably simplified.

Most spatial data from the EU-databases were obtained in 1-km grid resolution and the initial model construction therefore corresponded to this. The grid up-scaling was partly done automatically by the MIKE SHE model set-up-program, which uses the basic data files to establish a model with a larger user-specified grid resolution. However, manual adjustments were necessary in order to secure that the topographic representation matched the location of the river network while changing grid resolution and that the location of crop rotations matched the location of land use types.

Key results from simulations with different grid resolution are shown in *Table 3* and *Table 4*, and in *Figure 8*. The results show that the simulated annual runoff is almost identical and thus independent of grid sizes. An explanation of the minor differences is that the catchment areas in the 1, 2 and 4 km models were not exactly identical. Thus, the root zone processes responsible for generating the evapotranspiration and consequently the runoff do not appear to be scale dependent as long as the statistical properties of the soil and vegetation types are preserved, which is the case with the applied aggregation procedure.

*Table 3. Key results from simulation of the Karup catchment (518 $km^2$) with varying grid resolution (1, 2 and 4 km). Averaged results over the five-year period 1989-1993. Average annual precipitation was 884-mm $year^{-1}$.*

| | 1 km | 2 km | 4 km | Observed |
|---|---|---|---|---|
| River flow (mm $year^{-1}$) | 460 | 461 | 444 | 451 |
| % of area > 50 mg $NO_3$ $l^{-1}$ | 50.4 | 55.5 | 51.6 | 57 |
| Av. Conc. (mg $NO_3$ $l^{-1}$) | 45.5 | 47.2 | 47.3 | 58 |

*Table 4 Key results from simulation of the Odense catchment (536 $km^2$) with varying grid resolution (1, 2, and 4-km). Averaged results over the five-year period 1989-1993. Average annual precipitation was 805-mm $year^{-1}$.*

| | 1 km | 2 km | 4 km | Observed |
|---|---|---|---|---|
| River flow (mm $year^{-1}$) | 305 | 291 | 315 | 259 |
| % of area > 50 mg $NO_3$ $l^{-1}$ | 9.5 | 4.8 | 4.25 | 0 |
| Av. Conc. (mg $NO_3$ $l^{-1}$) | 24.5 | 17.9 | 15.3 | 2 |

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The hydrograph shape differed significantly for the three grid sizes (not shown). For the Karup model, the simulation with 1 km grid reproduced the low flow conditions reasonably well, whereas the 2 and 4 km grids had a rather poor description of the baseflow recession in general and the low flow conditions in particular. For the Odense model, the simulation with the 1 km grid showed too large baseflows during the low flow season, while the 2 km grid model obtained an appropriate level and the 4 km grid model simulated less low flow than observed. This indicates that there are significant effects of grid size on the stream-aquifer interaction that are not properly described in the applied aggregation procedure.

The simulated nitrate concentrations in the groundwater differ considerably between the two catchments. Higher average concentrations are simulated in the Karup catchments compared to the Odense catchment (*Table 3* and *Table 4*). For the Karup catchment the simulated nitrate distributions (*Figure 8*) compare rather well with measured data in terms of both shape and level. For the Odense catchment the simulated distribution of nitrate concentrations do not compare very well with the measured data. 80 % of the observation wells of the Odense catchment showed no nitrate whereas the model simulated zero concentration in only 25 % of the area. Thus, the simulated nitrate concentrations in the groundwater were not clearly influenced by the grid size for the Karup catchment, while there appeared to be some effect for the Odense catchment.

The explanation of these differences is found to the different hydro-geological situations in the two catchments. In the Karup catchment the groundwater table is generally located a couple of meters below terrain surface and the horizontal flows take place in both the Quaternary and the Miocene sediments. Hence, independent of grid resolution, the main part of the horizontal groundwater flow takes place in the about 15 m of the aquifer located above the reduction front. Only a relatively small part of the flow lines cross the reduction front, below which, the nitrate is assumed to disappear. In the Odense catchment, the horizontal groundwater flows take place almost exclusively in the lower aquifer, of which only the upper 3 metres are located above the reduction front. This implies that a large part of the groundwater flow is crossing the reduction front on its route from the infiltration zones in the hilly areas towards the discharge zones near the river. As the size of the grid influences the smoothness of the aquifer geometry, the grid size will significantly influence the number of flow lines crossing the reduction front and hence the nitrate concentrations. This scaling effect of the hydro-geological conditions is not accounted for in the present aggregation procedure.

It is noticed that the nitrate concentrations are significantly lower in the Odense catchment compared to the Karup catchment, with respect to both the observed and the simulated values. The main reason for this is that the different soil properties and the smaller number of animals result in a lower nitrate leaching from the root zone in the Odense catchment.

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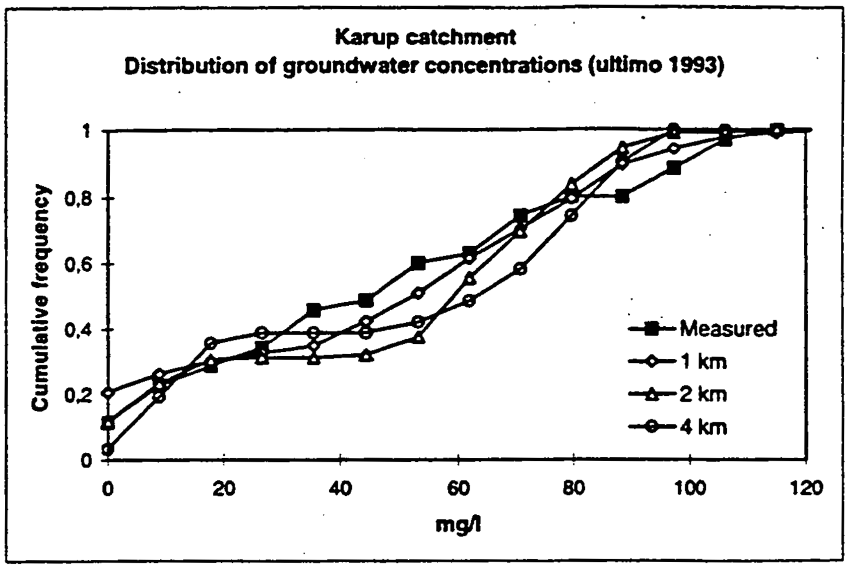


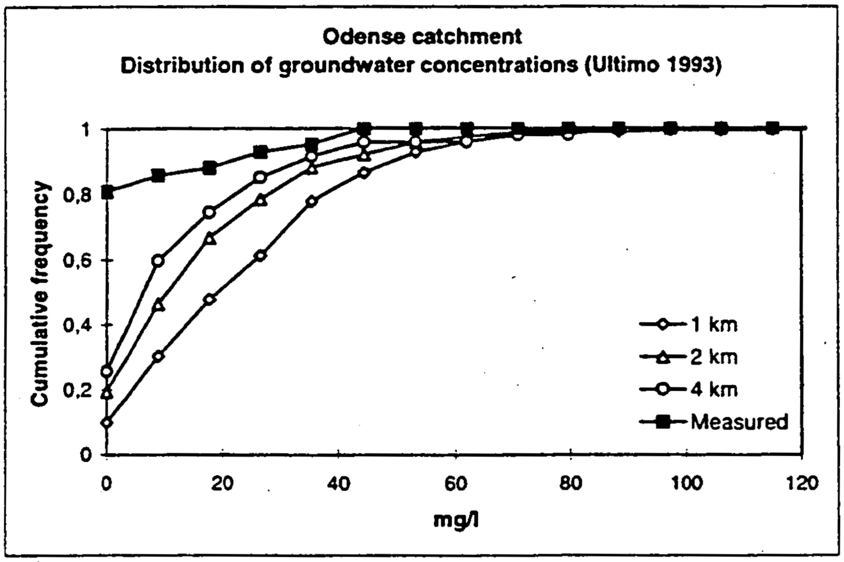


*Figure 8 Measured and simulated distributions of nitrate concentrations in upper groundwater in the Karup and the Odense stream catchments at the end of the simulation period. Measured distributions are based on 35 obervations in the Karup catchment and 42 observations in the Odense catchment.*

## 4.4 Uncertainty of Input Data

A simple way to compare effects of input data would be to simulate the same area based on two different datasets, generated from different sources. As may be seen from *Table 2*, the Karup catchment has been simulated with different data sources, but unfortunately not for the same period of time. The hydrology of the Odense catchment has been simulated twice: a) by applying local data (the client being the County of Funen) or b) with data from the EU-databases (in an EU-project). It is evident that the local data produce much more convincing

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results compared to the use of EU-databases as data source. A primary explanation of this discrepancy is the much better geological description obtained with local data.

Effects of input uncertainty may be judged through Monte Carlo simulations.
In the above-mentioned EU project, the uncertainty in simulated nitrate concentrations in groundwater aquifers was estimated using European level data sources readily available from standard European databases such as GISCO and EUROSTAT as the basis of the modelling. Where crucial data were found missing these were obtained from readily available national data sources. The model parameters were all assessed from these data by use of various transfer functions and no model calibration was carried out. Furthermore, a statistically based aggregation procedure, preserving the spatial distribution of soil types, vegetation types, etc. on a catchment basis was adopted.

The study area used for the uncertainty analysis was the Karup catchment, which is further described in Styczen and Storm (1993a,b). The data used for the study and the model construction are described in details in UNCERSDSS (1998), Refsgaard et al. (1999) and Hansen et al. (1999). The grid resolution used for the uncertainty analysis was 2 km. In the simulations, the DAISY model produced calculations of water and nitrogen behaviour from soil surface and through the root zone acting as source input to the MIKE SHE grids, which were allocated to agriculture. In the grids allocated to natural areas, MIKE SHE calculated the water percolation whereas no nitrate contribution was assumed from these areas. As the integration between DAISY and MIKE SHE was not made at code level when this project was conducted (*Figure 1*), feedback from MIKE SHE to DAISY through fluctuating groundwater was not considered. A seven-year period was simulated (1987-1993) of which the two first years were used for model warm-up.

Uncertainty analyses were performed using Monte Carlo technique where the deterministic model was run several times using different (equally probable) realisations of the input/parameter field. Ideally, all model parameters should be included in the uncertainty analysis. However, the MIKE SHE DAISY modelling system contain a very large number of input parameters, which, if all were to be treated stochastically, would require an unrealistic number of Monte Carlo simulations and CPU-time. Thus, a selection of only five types of model parameters was made using expert judgement to evaluate the expected relative influence of different model parameters on the type of model results, which were of particular interest in the present study, namely the nitrate concentrations of the aquifer. The magnitude of the error associated with each of the five stochastic parameters was, if possible, determined on the basis of information on the uncertainty related to the data source. If no information existed, the magnitude of the uncertainty was estimated by expert judgement. After evaluation of a number of test simulations, a total of 25 Monte Carlo runs were found adequate for the uncertainty analysis. The five stochastic parameters were

- Daily precipitation amount. The constraints were to keep dry days dry and to normalise series in order to preserve the mean value over the 25 Monte Carlo runs.
- Soil hydraulic properties determined by pedo-transfer functions based on soil textural composition. The clay content within each texture class of the GISCO soil map was treated stochastically.
- Slurry composition
- Soil organic matter content.
- Location of nitrate reduction front in the aquifer.

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Most of the stochastic parameters were related to the DAISY model. DAISY was not run in each grid cell in the catchment but only for each of 17 crop rotation types in order to limit the CPU time consumption. Thus, the stochastic parameters were not treated as spatially varying, but rather as spatially constant values. All stochastic parameters were treated as being mutually independent (uncorrelated). The reason for this was that the information for estimating correlation between the stochastic parameters was lacking and that no high degree of correlation was suspected a priori.

The key results from the Monte Carlo runs are shown in *Table 5*. The results are averaged over the five-year output period and the mean and standard deviations represent statistical values from the 25 Monte Carlo runs. The uncertainties on the simulated average groundwater concentrations are further shown in *Figure 9*.

*Table 5 Key results from 25 Monte Carlo runs. Simulation results are averaged for the entire catchment area and over the 5 year simulation period.*

| Variable | Mean | St.dev. | CV* (%) |
|---|---|---|---|
| Leaching from the root zone (kg ha$^{-1}$ year$^{-1}$) | 65 | 19 | 29 |
| Groundwater concentration (mg l$^{-1}$) | 48 | 8 | 17 |
| River flow (mm year$^{-1}$) | 464 | 22 | 5 |

* Coefficient of variation

Generally, less uncertainty was associated with the water balance component and river flow (CV=5 %) than to the components of the nitrogen balance i.e. nitrate leaching (CV=29 %) and nitrate concentrations in groundwater (CV=17 %) (*Table 5*). The uncertainty of simulated river flow was dominated by contributions from uncertainty on soil texture (influencing soil hydraulic parameters) and of precipitation, whereas the uncertainties associated with components of the nitrogen balance were dominated by the uncertainty contributions from both soil texture, soil organic matter and slurry composition.

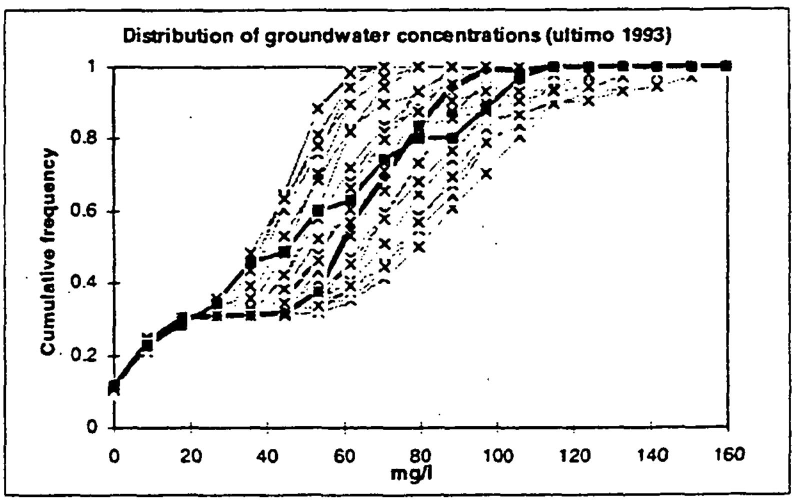


*Figure 9 Statistical distribution of groundwater nitrate concentrations in the Karup catchment by the end of the simulation period. . X = Simulated Monte Carlo distributions (25), ● = Deterministic simulation using 'best guess' parameters, ■ = measured distribution based on 35 groundwater boreholes.*

Uncertainty of precipitation contributed only to a minor degree to the simulated uncertainties of the nitrogen components despite the influence it had on the water balance. The depth of the reduction front appeared to have only minor influence on the uncertainty on stream water concentrations in the present simulations.

The case study indicates that, for the Karup catchment, the uncertainty of the predicted nitrate concentrations in the aquifer at a scale of some hundreds of $km^2$ is so relatively small that the methodology appears suitable for large scale policy studies. Results also show that even though the uncertainty of the simulated results is large at point/field scale, making the predictive capabilities questionable, it appears that the uncertainty at larger scales, where the point simulations are integrated in time and space, decrease considerably, making the simulation results appear more useful. Given the rather coarse data basis and given that no model calibration was performed, the simulated results of groundwater concentrations were remarkably good. Hence, the measured groundwater concentrations fell well within the predicted uncertainty intervals. As the Karup catchment is characterised as having flat topography and rather simple geology, it is most likely that the obtained results may differ for a catchment with a more varied topography and complex geology (see also Section 4.3).

## 5 Discussion and Conclusions

For simulation of water and nitrate, the models used have proven their worth at plot or field scale. It is therefore not surprising that it is possible to scale up area-wise, using the same, or almost the same grid resolution and level of detail in the data. A major leap occurs, when moving from "real" to statistical data. The experience from the different studies with nitrate simulations has been that this leap is indeed possible. However, a critical view on the evaluation of output is required. Due to the fact that the input is not geo-referenced, the comparison with output cannot be either. The model, therefore, cannot reproduce the concentration at a specific location in the catchment. It can represent the statistical distribution of concentrations in a given layer. Integrated outputs such as streamflow can also be simulated through the statistical approach.

It is interesting to notice that the distributed input (although based on statistics) is a requirement for a realistic result. In the UNCERSDSS project, an attempt was made to substitute the rotation approach by simulation of the dominant crop. The results were clearly erroneous. (*Figure 10*). The distribution of concentrations in the groundwater is very much a function of the interpretation of statistical data in terms of agricultural practice. This initial data interpretation therefore requires the necessary time and expertise.

For simulation of annual runoff and nitrate concentrations, both of which are affected primarily by root zone processes, the impact of changes of grid resolution is relatively small. Contrary to this, the impact on hydrograph shape is consistently rather large. This finding, which also is documented earlier in Refsgaard (1997), indicates that the applied aggregation procedure has important limitations with respect to describing the stream-aquifer interactions. An important point is that the applied methodology is scale dependent with regard to hydrograph simulation; hence a change of grid size generates a need for recalibration of parameters responsible for baseflow recession and low flow simulation.

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The Monte Carlo simulation exercise showed that it is possible to use Monte Carlo simulations to provide a good estimate of uncertainties in simulated groundwater concentrations if the basic model description is reliable. This makes simulation results more valuable for decision making.

Thus, the conclusion is that the data requirements seldom pose insurmountable problems for regional modelling, if the models used are considered reliable for the conditions to be investigated and if the aggregated results are sufficient for the decisions to be taken. For most political decision, it is the aggregated output, which are required. However, attention must be paid to the geology of the area, which is not adequately described in databases at EU level. National databases may, however, contain adequate data.

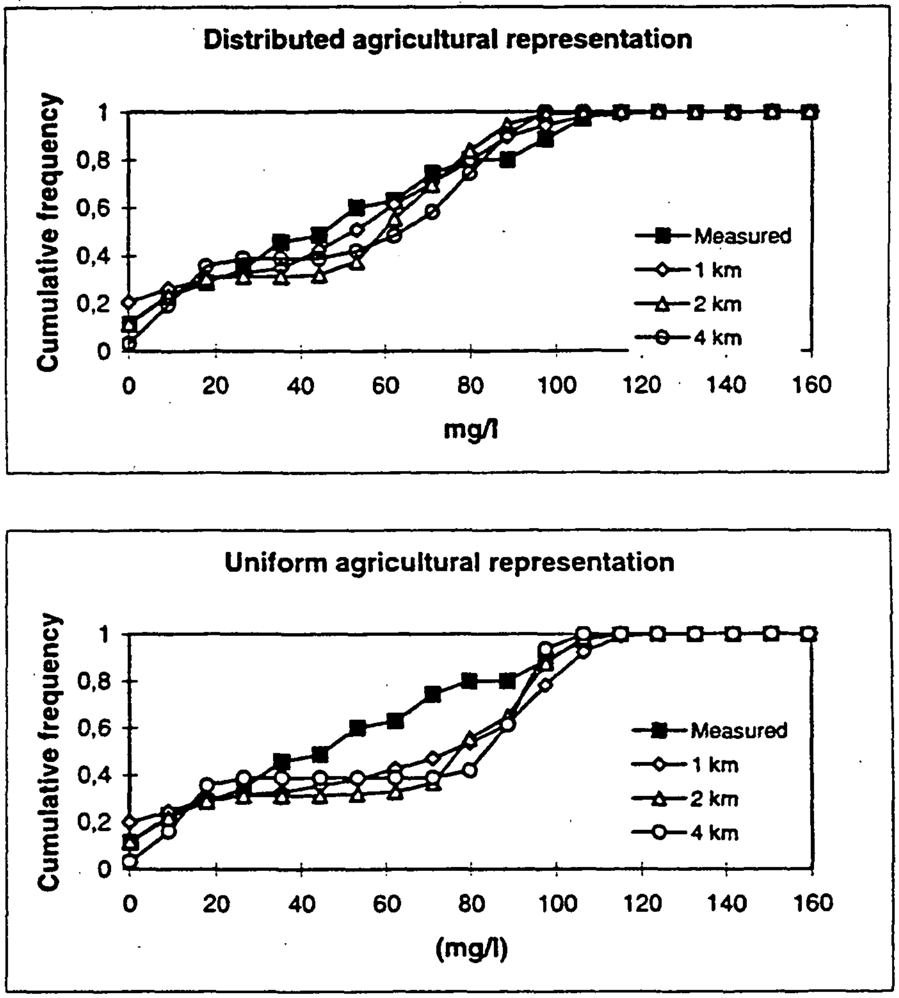


*Figure 10 Observed and simulated distribution of nitrate concentrations in the groundwater at the end of the simulation period (Karup catchment).*

For pesticides the picture is somewhat less clear at present. Generally, the pesticide models produce less convincing results than the other mentioned models, not least because the precision required of the simulations is very high. For comparison, the drinking water requirements state a concentration of less than 0.1 μg/l. If the pesticide application is 1 kg/ha and the application area is Eastern Denmark, it is necessary to precisely account for the last

1/5000 of the applied chemical. Heterogeneity may turn out to be of greater importance for such simulations than for e.g. nitrate. The results of the ongoing projects should indicate whether the statistical approach is valid also for this type of simulations.

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# Large scale modelling of groundwater contamination from nitrate leaching

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**Abstract**

Groundwater pollution from non-point sources, such as nitrate from agricultural activities, is a problem of increasing concern. Comprehensive modelling tools of the physically based type are well proven for small-scale applications with good data availability, such as plots or small experimental catchments. The two key problems related to large-scale simulation are data availability at the large scale and model upscaling/aggregation to represent conditions at larger scale. This paper presents a methodology and two case studies for large-scale simulation of aquifer contamination due to nitrate leaching. Readily available data from standard European level databases such as GISCO, EUROSTAT and the European Environment Agency (EEA) have been used as the basis of modelling. These data were supplemented by selected readily available data from national sources. The model parameters were all assessed from these data by use of various transfer functions, and no model calibration was carried out. The adopted upscaling procedure combines upscaling from point to field scale using effective parameters with a statistically based aggregation procedure from field to catchment scale, preserving the areal distribution of soil types, vegetation types and agricultural practices on a catchment basis. The methodology was tested on two Danish catchments with good simulation results on water balance and nitrate concentration distributions in groundwater. The upscaling/aggregation procedure appears to be applicable in many areas with regard to root zone processes such as runoff generation and nitrate leaching, while it has important limitations with regard to hydrograph shape due to its lack of accounting for scale effects in relation to stream aquifer interaction. 

*Keywords:* Upscaling; Databases; Non-point pollution; Nitrate leaching; Distributed model; Water balance

## 1. Introduction

Groundwater is a significant source of freshwater used by industry, agriculture and domestic users. However, increasing demand for water, increasing use of pesticides and fertilisers as well as atmospheric deposition constitute a threat to the quality of groundwater. The use of fertilisers and manure leads to the leaching of nitrates into the groundwater and atmospheric deposition contributes to the acidification of soils that may have an indirect effect on the contamination of water.

In Europe, for instance, the present situation is summarised in EEA (1995), where it is assessed that the major part of aquifers in Northern and Central Europe are subject to risk of nitrate contamination amongst others due to agricultural activities. Therefore, policy makers and legislators in EU are concerned about the issue and a number of preventive

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legislation steps are being taken in these years (EU Council of Ministers, 1991; EC. 1996).

In the scientific community, concerns on groundwater contamination have motivated the development of numerous simulation models for groundwater quality management. Groundwater models describing the flow and transport mechanisms of aquifers have been developed since the 1970s and applied in numerous pollution studies. They have mainly described the advection and dispersion of conservative solutes. More recently, geochemical and biochemical reactions have been included to simulate the transport and fate of pollutants from point sources as industrial and municipal waste disposal sites, see e.g. Mangold and Tsang (1991); Engesgaard et al. (1996) for overviews. Fewer attempts have been made to simulate non-point pollution at catchment scale resulting from agricultural activities, see e.g. Thorsen et al. (1996); Person et al. (1996) for overviews. The approaches range from relatively simple models with semi-empirical process descriptions of the lumped conceptual type such as ANSWERS (Beasley et al., 1980), CREAMS (Knisel, 1980; Knisel and Williams, 1995), GLEAMS (Leonard et al., 1987), SWRRB (Arnold and Wiliams, 1990; Arnold et al., 1995) and AGNPS (Young et al., 1995) to more complex models with a physically based process description. The physically based models are most commonly one-dimensional leaching models, such as RZWQM (DeCoursey et al., 1989, 1992), Daisy (Hansen et al., 1991) and WAVE (Vereecken et al., 1991; Vanclooster et al., 1994, 1995), which basically describe root zone processes only, while true, spatially distributed, catchment models based on comprehensive process descriptions, such as the coupled MIKE SHE/Daisy (Styczen and Storm, 1993), are seldom reported. The simple conceptual models are attractive because they require relatively less data, which are usually easily accessible, while the predictive capability of these models with regard to assessing the impacts of alternative agricultural practises is questionable due to the semi-empirical nature of the process descriptions. On the contrary, a key problem in using the more complex catchment models operationally lies in the generally large data requirements prescribed by the developers of such model codes. However, due to the better process descriptions these models may for some types of application be expected to have better predictiv capabilities than the simpler models (Heng and Niko laidis, 1998).

Input data for the complex catchment models hav traditionally been available in practise only for smal areas such as experimental research catchment However, as more and more data have been gathere in computerised databases and, in particular, i Geographical Information Systems (GIS), the dat availability has improved significantly. Furthe experience from case studies indicates that a consid erable part of the input data may be derived fron statistical data and more general databases (Stycze and Storm, 1995).

The database of EUROSTAT, the statistical offic of the European Commission, holds statistical in mation about different topics from all Member of the European Union. Agricultural statistics provid information on main crops, on the structure of agri cultural holdings and crop and on animal productior Environment statistics provide figures on impacts c other sector's work on the environment, such as ferti liser and pesticide input, groundwater withdrawa water quality or manure production on anima farms. These figures are mostly aggregated an published on national level.

In order to use these statistics in a spatially distrib uted simulation model, the information needs to b spatially referenced to represent a unit on the grounc Therefore the statistical information needs to b linked to a GIS data set. Such GIS data is stored i the GISCO (Geographic Information System of th European Commission) database. The GISCO base holds spatial data about administrative bound aries down to commune level, thematic data se such as the soil database, CORINE land cover (mana ged by the EEA) or climatic time series for about 200 measuring stations in the European Union.

Thus on one hand, there is a clearly expressed nee from decision makers at national and internation level to have tools, which on the basis of readily avai able data can predict the risks of groundwater poll tion from non-point sources and the impacts alternative agricultural management practices; an on the other hand, the scientific community h achieved new knowledge and developed new too aiming at this. However, there are some importa gaps to be filled before the scientifically based too

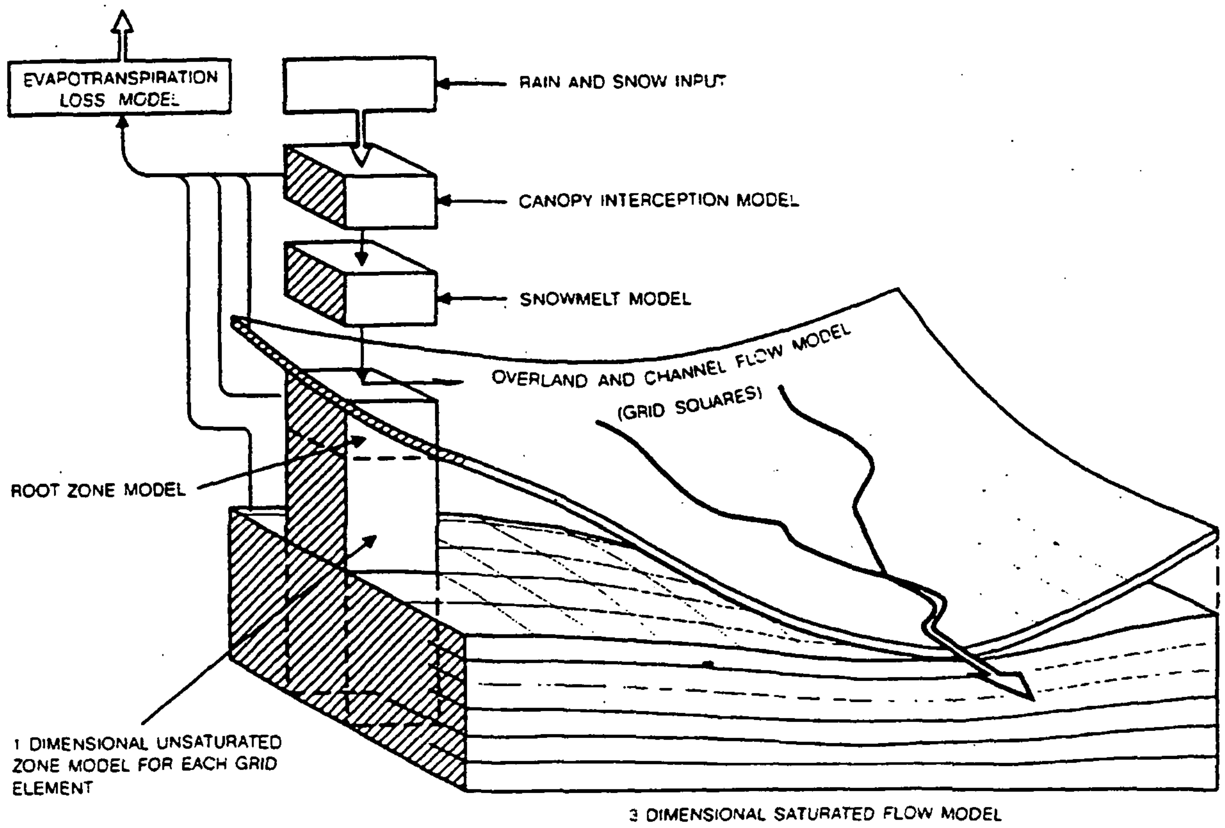


Fig. 1. Schematic structure of the MIKE SHE.

can be applied operationally for supporting the decision makers:

- The physically based models are very promising tools for assessing the impacts of alternative agricultural practises, but have so far been tested on plot scale and very small experimental catchments, whereas the need from a policy making point of view mainly relates to application on a much larger scale. Hence, there is a need to derive and test methodologies for upscaling of such models to run with model grid sizes one to two order of magnitudes larger than usually done.
- Readily available data on large (national and international) scales do exist, although in a somewhat aggregated form. However, such data have not yet been used as the basis for comprehensive modelling, which so far always have been based on more detailed data, often from experimental catchments. Hence, there is a need to test to which extent these readily available data are suitable for modelling.
- There is a need to assess the predictive uncertainties, before it can be evaluated whether the approach of combining complex predictive models with existing data bases is of any practical use in the decision making process or whether the uncertainties are too large.

This paper presents results from a joint EU research project on prediction of non-point nitrate contamination at catchment scale due to agricultural activities. Other results from the same study focussing on uncertainty aspects are presented in UNCERSDSS (1998), Refsgaard et al. (1998a, 1999) and Hansen et al. (1999).

## 2. Methodology

### 2.1. Materials and methods

#### 2.1.1. MIKE SHE

MIKE SHE is a modelling system describing the flow of water and solutes in a catchment in a distributed physically based way. This implies numerical

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solutions of the coupled partial differential equations for overland (2D) and channel flow (1D), unsaturated flow (1D) and saturated flow (3D) together with a description of evapotranspiration and snowmelt processes. The model structure is illustrated in Fig. 1. For further details reference is made to the literature (Abbott et al., 1986; Refsgaard and Storm, 1995).

#### 2.1.2. Daisy

Daisy (Hansen et al., 1991) is a one-dimensional physically based modelling tool for the simulation of crop production and water and nitrogen balance in the root zone. Daisy includes modules for description of evapotranspiration, soil water dynamics based on Richards' equation, water uptake by plants, soil temperature, soil mineral nitrogen dynamics based on the advection–dispersion equation, nitrate uptake by plants and nitrogen transformations in the soil. The nitrogen transformations simulated by Daisy are mineralization–immobilization turnover, nitrification and denitrification. In addition, Daisy includes a module for description of agricultural management practices. Details on the Daisy application in the present study are given by Hansen et al. (1999).

#### 2.1.3. MIKE SHE/Daisy coupling

By combining MIKE SHE and Daisy, a complete modelling system is available for the simulation of water and nitrate transport in an entire catchment. In the present case the coupling is a sequential one. Thus for all agricultural areas, Daisy first produces calculations of water and nitrogen behaviour from the soil surface and through the root zone. The percolation of water and nitrate at the bottom of the root zone simulated by Daisy, is then used as input to MIKE SHE calculations for the remaining part of the catchment. For natural areas, MIKE SHE calculates also the root zone processes assuming no nitrate contribution from these areas. Owing to the sequential execution of the two codes, it has to be assumed that there is no feed back from the groundwater zone (MIKE SHE) to the root zone (Daisy). Further, overland flow generated by high intensity rainfall (Hortonian) cannot be simulated by this coupling, while overland flow due to saturation from below (Dunne) can be accounted for by MIKE SHE.

Thus, MIKE SHE does not in the present case handle evapotranspiration and other root zone processes in the agricultural areas. As Daisy is or dimensional, one Daisy run in principle should carried out for each of MIKE SHE's horizon grids. However, several MIKE SHE grids are assum to have identical root zone properties (soil, crop, ag cultural management practices, etc.), so that in pra tise the outputs from each Daisy run can be used input to several MIKE SHE grids.

### 2.2. Data availability at European databases

Input data for modelling at the European scale ne to satisfy certain requirements to make them useful t large-scale applications:

- The data must be available for the whole Europe.
- The data must be harmonised according to common nomenclature in order to avoid regior or national inconsistencies.
- The data should be available in a seamless da base.
- The data should be available from one sing source to avoid regional or national inconsiste cies.
- The data should be available in a format which c be directly integrated into a Geographical Inform tion System (GIS).

Attached to the use of "European" data sets are al certain problems. The data are generalised geometric as well as in thematic detail, local partic larities which are especially important for hydrolo cal simulations are not always accounted for information that is required for specific modelli objectives is not directly available on Europe level demanding the establishment and use of transi functions instead. On the contrary, information sometimes too specific when it has been collected the framework of a particular research project, e information on a particular soil property is bei collected in natural soils but not in agricultural soi

Given these formal requirements, a first task of t project was to study the availability of data sets suit for large-scale hydrological modelling of groun water contamination from diffuse sources. Af intensive searches of on-line data catalogues, pap publications and direct contacts with organisatio holding relevant information, it was possible

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Table 1
Data sources for European scale hydrological modelling

| Data | Potential data source identified in European data base | Source actually used for modelling | Scale of available data used |
|---|---|---|---|
| Topography | USGS[a]/GISCO | USGS/GISCO | 1 km grid |
| Soil type | GISCO soil map | GISCO soil map | 1 km grid |
| Soil organic matter | RIVM[b] report | Experience value for Danish arable soils[c] | Denmark |
| Vegetation | EEA: CORINE land cover | EEA: CORINE land cover | 1 km grid |
| River network and river cross sections | DCW[d] | Provided by an application developed within the project | 1 km grid |
| Geology | Report on groundwater resources in Denmark (EC, 1982) RIVM—digital map data of report. | Report on groundwater resources in Denmark (EC, 1982) | County, i.e. approximately 3.000 $km^2$ |
| Groundwater abstraction | Report on groundwater resources in Denmark (EC, 1982) RIVM—digital map data of report | Report on groundwater resources in Denmark (EC, 1982) | Commune, i.e. approximately 200 $km^2$ |
| Management practices | SC-DLO[e] report | Plantedirektoratet (1996) | Denmark |
| Crop type | Eurostat—Regional Statistics | Agricultural Statistics (1995) | County, i.e. approximately 3000 $km^2$ |
| Livestock density | Eurostat—Regional Statistics Eurostat—Eurofarm | Agricultural Statistics (1995) | County, i.e. approximately 3000 $km^2$ |
| Fertilizer consumption | Eurostat—Environmental Statistics | Agricultural Statistics (1995) | County, i.e. approximately 3000 $km^2$ |
| Manure production | Eurostat—Environmental Statistics | Agricultural Statistics (1995) | County, i.e. approximately 3000 $km^2$ |
| Atmospheric deposition | MARS project | National data | Denmark |
| Climatic variables | MARS project[f] | National data | Denmark |
| River runoff | GRDC[g] | National data | Catchment |

[a] USGS—United States Geological Survey.
[b] RIVM—National Institute of Public Health and the Environment of The Netherlands.
[c] RIVM data only include natural areas, not arable land. Instead the figure was assessed on the basis of previous experience with Danish agricultural soils.
[d] DCW—Digital Chart of the World.
[e] SC-DLO—Winand Staring Centre, The Netherlands.
[f] MARS—Monitoring Agriculture by Remote Sensing database.
[g] GRDC—Global Runoff Data Centre, database mainly for large river basins.

identify sources for all the information requirements. However, after evaluation of all the potential sources the following deficiencies became apparent:

- Not all information was available in spatially referenced GIS format, therefore other sources such as tables and statistics had to be considered.
- Not all information was available from "European" databases, finally national sources had to be considered. For these national sources strict requirements in terms of ease of availability, data quality and data comparability were imposed.
- The scale of the available data was often too coarse for the application. Global data sets with $1 \times 1°$ longitude/latitude resolution are often not detailed enough.

The potential "European scale" data sources and the data sources which ultimately was used for the model are shown in Table 1.

Data about climatic variables were obtained from the national meteorological institutes and river runoff from the national hydrological institutes. These data were only available from national sources, but on the contrary these data are probably the most easily available (if the issue of price charges is disregarded) and the most easily comparable due to international harmonised measuring techniques at these organisations. Regional statistics on Denmark obtained from EUROSTAT proved to be not detailed enough (country level only). The required statistical information could easily be recovered from Danish national statistics.

Cost estimates for the compilation of the database have only been undertaken to a limited extent. The project data itself have mostly been obtained in exchange for the anticipated project results, i.e. at no cost. The main data that in a fully commercial environment cost a substantial amount of money are meteorological data which are available from the national meteorological institutes (Kleeschulte, 1998).

### 2.3. Change of scale

Large scale hydrological models are required for a variety of applications in hydrological, environmental and land surface-atmosphere studies, both for research and for day to day water resources management purposes. The physically based models have so far mainly been tested and applied at small scale and therefore require upscaling. The complex interactions between spatial scale and spatial variability is widely perceived as a substantial obstacle to progress in this respect (Blöschl and Sivapalan, 1995; and many others).

The research results on the scaling issue reported during the past decade have, depending on the particular applications, focussed on different aspects, which may be categorised as follows:

- *Subsurface processes* focussing on the effect of geological heterogeneity.
- *Root zone processes* including interactions between land surface and atmospheric processes.
- *Surface water processes* focussing on topographic effects and stream–aquifer interactions.

The effect of spatial heterogeneity on the description of subsurface processes has been the subject of comprehensive research for two decades, see e.g. Dagan (1986) and Gelhar (1986) for some of the first consolidated results and Wen and Gómez-Hernández (1996) for a more recent review, mainly related to aquifer systems. The focus in this area is largely concerned with upscaling of hydraulic conductivity and its implications on solute transport and dispersion processes in the unsaturated zone and aquifer system, typically at length scales less than 1 km.

The research in the land surface processes has mainly been driven by climate change research where the meteorologists typically focus on length scales up to 100 km. Michaud and Shuttelworth (1997), in a recent overview, conclude that substantial progress has been made for the description of su[illegible] energy fluxes by using simple aggregation rules. Sellers et al. (1997) conclude that "it appears that simple averages of topographic slope and vegetation parameters can be used to calculate surface energy and heat fluxes over a wide range of spatial scales, from a few meters up to many kilometers at least for grassland and sites with moderate topography". An interesting finding is the apparent existence of a threshold scale, or representative elementary area (REA) for evapotranspiration and runoff generation processes (Wood et al., 1988, 1990, 1995). Famiglietti and Wood (1995) concludes on the implications of such an REA in a study of catchment evapotranspiration that "the existence of an REA for evapotranspiration modelling suggests that in catchment areas smaller than this threshold scale, actual patterns of model parameters and inputs may be important fa[illegible] governing catchment-scale evapotranspiration rates in hydrological models. In models applied at scales greater than the REA scale, spatial patterns of dominant process controls can be represented by their statistical distribution functions". The REA scales reported in the literature are in the order of 1–5 $km^2$.

The research on scale effects related to topography and stream–aquifer interactions has been rather limited as compared to the above two areas. Saulnier et al. (1997) have examined the effect of the grid sizes in digital terrain maps (DTM) on the model simulations using the topography-based TOPMODEL. They concluded that in particular for channel pixels the spatial resolution of the underlying DTM is important. Refsgaard (1997) using the distributed MIKE SHE

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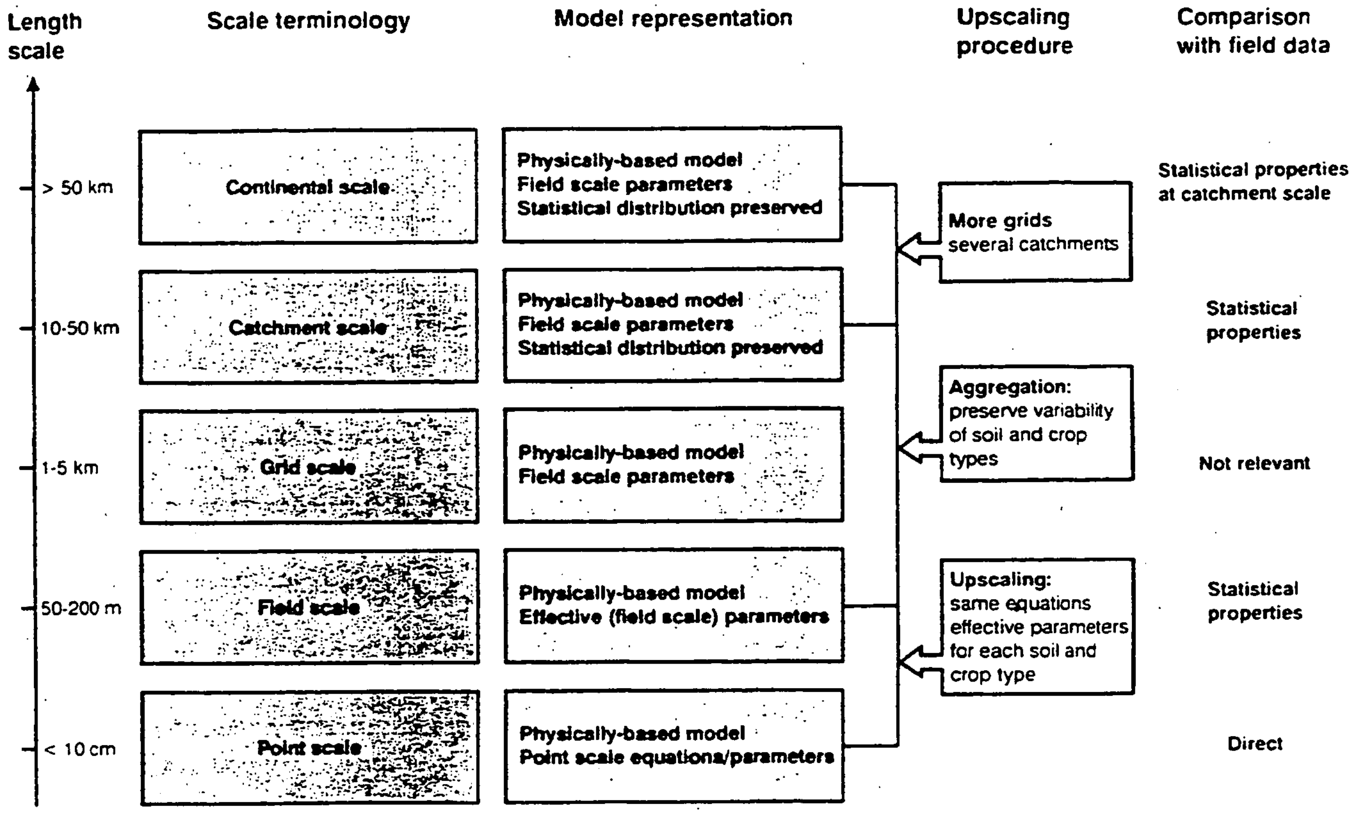


Fig. 2. Schematic representation of upscaling/aggregation procedure.

model to the Danish Karup catchment with grid sizes of 0.5, 1, 2 and 4 km, found that the discharge hydrograph shape was significantly affected for the 2 and 4 km grids as compared to the almost identical model results with 0.5 and 1 km grids. He concluded that the main reason for this change was that the density of smaller tributaries within the catchment was smaller for the models with the larger grids.

Many researchers doubt whether it is feasible to use the same model process descriptions at different scales. For instance Beven (1995) states that "... the aggregation approach towards macroscale hydrological modelling, in which it is assumed that a model applicable at small scales can be applied at larger scales using 'effective' parameter values, is an inadequate approach to the scale problem. It is also unlikely in the future that any general scaling theory can be developed due to the dependence of hydrological systems on historical and geological perturbations". We have experienced some of the same problems and agree that it is generally not possible to apply the same model without recalibration at small and large scales. Therefore, we have used another approach based on a combination of aggregation and upscaling in accordance with the principles recommended by Heuvelink and Pebesma (1998). The scale terminology and the upscaling procedure adopted here are as follows (Fig. 2):

- The basic modelling system is of the distributed physically based type. For application at *point scale* (where it is not used spatially distributed) the process descriptions of this model type can be tested directly against field data.
- The model is in this case run with (equations and) parameter values in each horizontal grid point representing *field scale* (50–200 m) conditions. The field scale is characterised by 'effective' soil and vegetation parameters, but assuming only one soil type and one cropping pattern. Thus the spatial variability within a typical field is aggregated and accounted for in the 'effective' parameter values.
- The smallest horizontal discretization in the model is the *grid scale* or grid size (1–5 km) that is larger than the field scale. This implies that all the variations between categories of soil type and crop type



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Fig. 3. Locations of the Karup and Odense catchments in Denmark.

within the area of each grid cannot be resolved and described at the grid level. Such input data whose variations are not included in the grid scale model representation, are distributed randomly at the catchment scale so that their statistical distributions are preserved at that scale.

- The results from the grid scale modelling are then aggregated to *catchment scale* (10–50 km) and the statistical properties of model output and field data are then compared at catchment scale.
- For applications to larger scales than catchment scale, such as *continental scale*, the catchment scale concept is used, just with more grid points. This implies that the continental scale can be considered to consist of several catchments, within each of which the field scale statistical variations are preserved and at which scale the predictive capability of the model thus lies.

In the upscaling procedure a distinction is made between the terms *upscaling* and *aggregation*. Thus, spatial attributes are aggregated and model parameters are scaled up. A principal difference between aggregation and upscaling is that whereas aggregation can be defined irrespective of a model operating on the aggregated values, upscaling must always be defined in the context of a model that uses the parameters that have been scaled up (Heuvelink and Pebesma, 1998). In this respect the main principle of the upscaling procedure can be summarised as follows:

- Upscale model from point scale to field scale.
- Run model at grid scale using field scale parameters in such a way that their statistical properties are preserved at catchment scale.
- Aggregate grid scale model output to catchment scale.

This methodology mainly attempts to address scaling within the second of the above fields, namely root zone processes, while scaling in relation to subsurface processes and stream–aquifer interaction has not been considered when designing the present upscaling procedure. The methodology has some complications and critical assumptions:

- The assumption of upscaling from point scale to field scale is crucial. This assumption is documented to be fulfilled in many cases (Jensen and Refsgaard 1991a–c; Djuurhus et al., 1999), but may fail in other cases (Bresler and Dagan, 1983), for instance in areas where overland flow is a dominant flow mechanism.
- Running the model at grid scale but using model parameters valid at a field scale, which is typically 2 to 3 orders of magnitude smaller, is necessary to make the computational demand acceptable for catchment and continental scale applications. The solution to this is to assign inputs on soil and vegetation types not correctly georeferenced but such that their statistical distribution at catchment scale is preserved. This implies that results at grid scale are dubious and should not be used. The aggregation step up to catchment scale is therefore essential.
- While the statistical properties of the critical root zone parameters due to the aggregation step have been preserved at catchment scale this is not the case for the geological, topographical and stream data which are used directly at the grid scale. A critical question is therefore, how the catchment scale model output, due to these other data, are influenced by selection of grid scale. Here, investigations with 1, 2 and 4 km grids are made.

## 3. Application

### 3.1. Modelling approach for the Karup and Odense catchments

The modelling studies have focussed on two aspects, namely the feasibility of using coarse aggregated data available at European level databases, and the effect of the upscaling procedure. The modelling aims at describing the integrated runoff at the catchment outlet and the distribution function of the nitrate concentrations sampled from available wells over the catchment (aquifer). On this basis the following approach has been adopted:

1. Simulation models have been established for two catchments in Denmark, Karup Å and Odense Å (Fig. 3), in the following denoted the Karup and Odense models, respectively. The topographical areas for the Karup catchment gauging station 20.05 Hagebro is 518 km$^2$. Correspondingly, the catchment area at the gauging station used for the model validation tests in the Odense catchment, 45.26 Ejby Mølle, is 536 km$^2$. The most detailed studies were carried out for the Karup catchment, while the results for the Odense catchment were included mainly to check the generality of the conclusions derived from the Karup catchment.
2. The models are established directly from the European level databases and all input parameter values are assessed from these data or in a predefined objective way from experience values obtained from previous model studies. Thus, the models are not calibrated at all.
3. The results of the models are compared with field data, on which basis the model performance is assessed.
4. The effects of upscaling have been examined in two ways:
   - The models are run with different grid sizes (1, 2 and 4 km) and the results compared.
   - For the Karup catchment two different procedures have been compared, namely:

     the upscaling/aggregation procedure described above (Fig. 2), which according to its representation of agricultural crops is denoted 'distributed';

     a simpler procedure where the agricultural crops are upscaled all the way from field scale to catchment scale. This implies that one crop type represents all the agricultural areas. The dominant crop in the area, namely winter wheat, has been selected as the crop for the 70% agricultural area, while the 30% natural/

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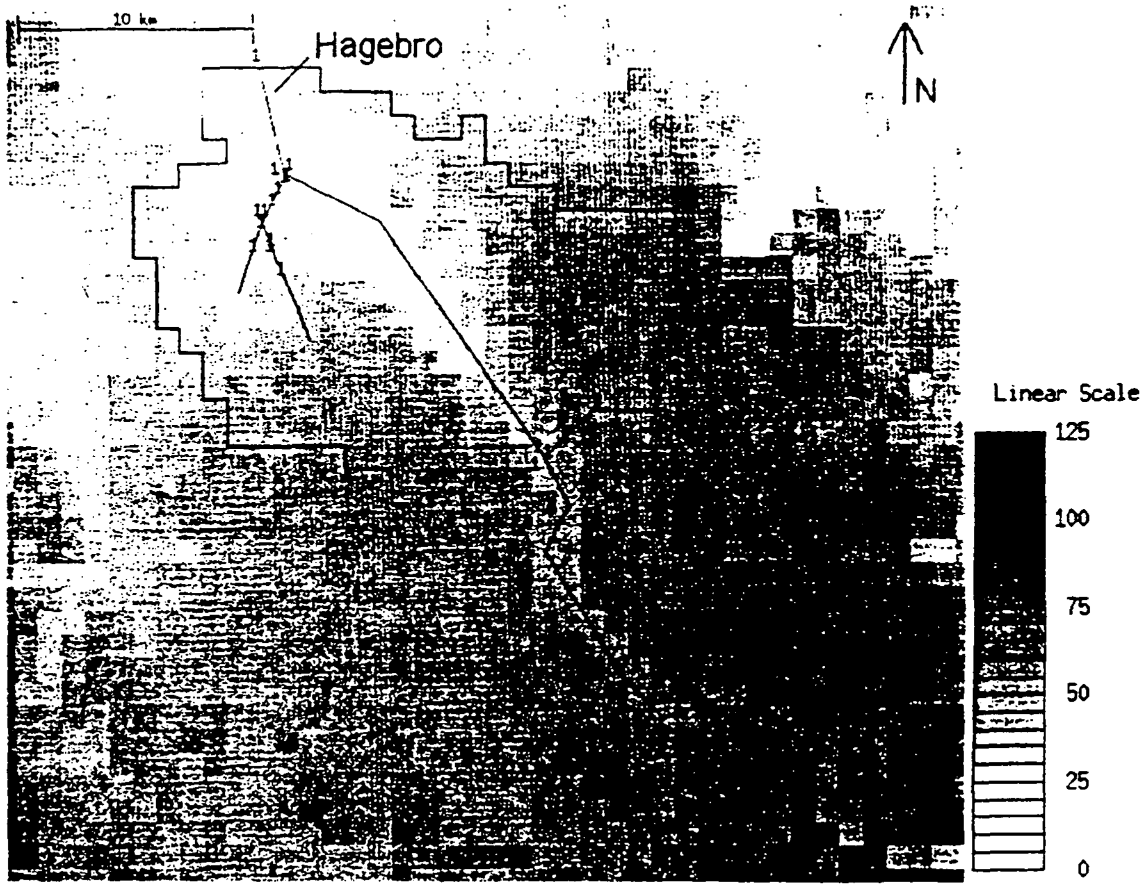


Fig. 4. Surface topography, catchment delineation and river network for the Karup-EU model.

urban areas remain as the only other vegetation type. This procedure is denoted 'uniform'.

### 3.2. Karup model

#### 3.2.1. Catchment and river system

The catchment area and locations of the river branches (Fig. 4) were generated from the DEM by use of standard ARC/Info functionalities. The generated catchment areas for 1, 2 and 4 km grids were within 4% of the correct one at station 20.05 Hagebro. The river cross-sections were subsequently automatically derived on the basis of the following assumptions:

- The bankful discharge (i.e. water flow up to top of cross-section) corresponds to a typical annual maximum discharge. This characteristic discharge is further assumed uniform in terms of specific runoff ($l\ s^{-1}\ km^{-2}$), so that the actual discharge at any cross section is estimated as the specific runoff multiplied by the upstream catchment area that can be estimated from the DEM.
- The river slope corresponds to the slope of the surrounding surface, which can be derived from the DEM.
- The cross-section has a trapezium shape w fixed given angle and relation between depth and width.
- The relation between discharge, slope and river cross-section can be determined by the Manning formula with a given Manning number.

Most areas in Denmark are drained in order to make the land suitable for agriculture. Agricultural areas are typically artificially drained with tile drains in combination with small ditches. Other areas may be naturally drained by creeks and rivers. It is not possible to include a detailed and fully correct drainage description in a coarse model like the Karup model. Moreover, detailed information on drainage network is not available. Therefore, when establishing a coarse scale model, a lumped description must be used. In the present case it is simply assumed that the entire catchment area is drained and that the drains are located 1 m below ground surface. Drainage water is produced whenever the groundwater table is located above this drainage level. Drainage water is routed to the nearest river node where it contributes as a source to the river flow. Routing of groundwater to the drains and further to the ultimate recipient is in MIKE SHE described using a linear routing technique, where a time constant is specified by the user. In this case a time constant of $2.3 \times 10^{-7}\ s^{-1}$ was used corresponding to an average retention time (in the linear reservoir) of 50 days. This time constant represents a typical value for Danish catchments.

### 3.2.2. Soil properties

The soil texture classes in a $1 \times 1$ km resolution were provided by the GISCO soil data base. The texture classes were translated into soil parameters in terms of hydraulic conductivity functions and soil water retention curves using pedo-transfer functions (Cosby et al., 1984). According to the GISCO the Karup catchment is covered by coarse sandy soil for which the following key parameter values were estimated: (a) saturated hydraulic conductivity $K_s = 1.7 \times 10^{-5}$ m/s; (b) moisture content at saturation $\theta_s = 40$ vol%; (c) moisture content at field capacity $\theta_{FC} = 20$ vol%; and (d) moisture content at wilting point $\theta_{wp} = 6$ vol%.

A specific problem was related to assessment of soil organic matter, which is an important parameter for nitrogen turnover processes. As indicated in Table 1 such information was not identified in any of the European data bases. Instead a value based on previous experience (Lamm, 1971) with Danish agricultural soils was estimated. In the plough layer (0–20 cm) a value of 1.5%C was used, and this value decreased rapidly with depth to a minimum of 0.01%C below 1 m depth.

### 3.2.3. Hydrogeology

The geological perception of the area and the basis for estimation of the hydrogeological parameters used in the model are all based on EC (1982), where the aquifer is described as composed of two main geological layers.

The upper layer is Quaternary sediments consisting of sands and gravel. The transmissivity of these sediments are assessed to be in the order of $2 \times 10^{-3}\ m^2/s$ and the thickness about 15 m (EC, 1982). This leads to a horizontal hydraulic conductivity of $1.3 \times 10^{-4}$ m/s that was used in the model calculations. An anisotropy factor of 10 between horizontal and vertical hydraulic conductivities was assumed leading to a vertical hydraulic conductivity of $1.3 \times 10^{-5}$ m/s. Moreover, a specific yield of 0.2 and a storage coefficient of $10^{-4}\ m^{-1}$ was assumed.

Below the Quaternary sediments there are Miocene quarts-sand sediments with a relatively high transmissivity of $3 \times 10^{-3}\ m^2/s$ and a thickness of typically 10–20 m (EC, 1982). Hence, in the model a thickness of 15 m has been used. This leads to a horizontal hydraulic conductivity of $2.0 \times 10^{-4}$ m/s. The same assumptions on anisotropy, specific yield and storage coefficients as for the Quaternary sediments were applied for the Miocene sediments.

EC (1982) provides information on groundwater abstraction on a commune (local administrative unit) basis. The Miocene sediments are described as suitable for drinking water supply, why it is assumed that all groundwater abstractions are made from these sediments that are the lower layer in the model. The total abstraction is given as $13 \times 10^6\ m^3$/year. The exact location of the individual water supply wells is not given in EC (1982), and has been evenly distributed among 10–20 model grids located along the river system.

The location of the reduction front in the aquifer is an important parameter for nitrate conditions. As percolation water containing nitrate moves into areas with reduced geochemical conditions the nitrate will disappear. No information on this important parameter was provided in EC (1982). It was assumed that the front separating oxic and reduced aquifer conditions all over the aquifer is located in the Miocene sediments, 3 m below the interface to the Quaternary sediments. This corresponds to a location 18 m below the terrain surface.

### 3.2.4. Hydrometeorology

Time series of daily precipitation and temperature based on standard meteorological stations within the catchment was used. In addition, monthly values of potential evapotranspiration were calculated by the Makkink equation on the basis of climate data from the synoptic station at Karup airport. The data from synoptic stations are generally easily available internationally.

### 3.2.5. Crop growth, evapotranspiration and nitrate leaching model

Distributions of crop types and livestock densities were obtained from Agricultural Statistics (1995) and converted to slurry production using standard values for nitrogen content. Based on typical crop rotations proposed by The Danish Agricultural Advisory Centre and the constraints offered by crop distribution and livestock density two cattle farm rotations, one pig farm rotation and one arable farm rotation were constructed. In order to capture the effect of the interaction between weather conditions and crops, simulations were performed in such a way that each crop at its particular position in the considered rotation occurred exactly once in each of the years, which resulted in a total of 17 crop rotation schemes. These 17 schemes were distributed randomly over the area in such a way that the statistical distribution was in accordance with the agricultural statistics.

To simulate the trend in the nitrate concentrations in the groundwater and in the streams, it is necessary to have information on the history of the fertiliser application in space and time. In Denmark, norms and regulations for fertilisation practice are defined (Plantedirektoratet, 1996) which regulate the maximum amount of nutrients allowed for a particular crop depending on fore-fruit and soil type, and in addition, provide norms for the lower limit of nitrogen utilisation for organic fertilisers. It was assumed that the farmers follow the statuary norms, and that the proportion of organic fertiliser to the individual crop in a rotation is proportional to the production of organic fertiliser in the rotation and to the relative nitrogen demand of the crop (the fertiliser norm of the particular crop in relation to the fertiliser norm of the rotation). Based on estimated application rates of organic and mineral fertilisers to the individual crops each year, the Daisy model simulated time series of nitrate leaching from the root zone for each agricultural grid. The MIKE SHE model then routed these fluxes further through the unsaturated zone and in the groundwater layers accounting for dispersion and dilution processes and finally into the Karup stream where the integrated load from the entire catchment was estimated.

The parameterisation of the Daisy model is adopted from previous studies. The basic processes and standard parameter values were originally assessed from results of Danish agricultural field experiments (Hansen et al., 1990). As then the process description and standard parameters have only been subject to minor modifications in connection with model tests against data from The Netherlands, Germany, Denmark and Slovakia (Hansen et al 1991; Jensen et al. 1994 1996, 1997; Svendsen et al, 1995). Hence, the parameters related to both, evapotranspiration water balance processes and to the nitrogen transformation processes have, except for the soil parameters described in Section 3.2.3, been taken as the standard values. More details on the parameter values, their assessed uncertainties and results from the Daisy simulations are provided in Hansen et al. (1999).

### 3.2.6. Boundary and initial conditions

In addition to precipitation and groundwater abstraction rates the following boundary conditions are used:

- The area included in the catchment is per definition a hydrological catchment as based on topography Thus a zero-flux boundary is used along the catchment boundaries, also for the aquifer layers. The bottom of the model is considered impermeable
- For all upstream river ends a zero-flux boundary condition is applied. For the downstream end, a constant water level was applied.

The most important initial conditions are the moisture content in the unsaturated zone and the elevation of the groundwater table. The initial soil moisture content was assumed equal to field capacity, while the initial groundwater tables was assumed equal to the groundwater tables after a seven years simulation period with guessed initial conditions. The model was run for seven years (1987–1993). In order to reduce the importance of uncertain initial conditions, the two first years were considered as a 'warming-up period' and the last five years were considered the simulation period.

Table 2
Water balance in mm/year for the Karup catchment at station 20.05 Hagebro (518 km$^2$)

| Year | Precipitation | River flow | | | Observed |
|---|---|---|---|---|---|
| | | Model 1 km grid | Model 2 km grid | Model 4 km grid | |
| 1989 | 812 | 428 | 392 | 353 | 460 |
| 1990 | 1020 | 496 | 518 | 512 | 476 |
| 1991 | 863 | 446 | 441 | 424 | 449 |
| 1992 | 892 | 499 | 531 | 527 | 437 |
| 1993 | 835 | 434 | 425 | 405 | 432 |
| Average | 884 | 460 | 461 | 444 | 451 |

### 3.3. Odense model

The same procedure as outlined above for the Karup model was followed. The two main differences as compared to the Karup catchment are that the top soil belong to more fine textured classes with lower hydraulic conductivities and that the aquifer having groundwater abstraction is confined in the Odense catchment. This results in an assumption that the covering sediments are less permeable than the aquifer material. As no direct information on these confining sediments is given in EC (1982) the hydraulic properties of the soil in the root zone are assumed valid. This implies in practise that recharge rates to the aquifer is lower than in the Karup catchment and that the horizontal flow towards the drains and the river system is correspondingly larger. A similar geological geometry as in the Karup catchment is assumed, i.e. the upper less permeable, confining layer is assumed to have a thickness of 15 m and the reduction front is assumed to be located in the lower aquifer, 3 m below this confining layer.

## 4. Results

To test the model performance a number of validation tests were carried out for both catchments. Validation is here defined as substantiation that a site specific model performs simulations at a satisfactory level of accuracy. Hence, no universal validity of the general model code is tested nor claimed. In Tables 2 and 3 and Figs. 5–8 results are shown for model grid sizes 1, 2 and 4 km and for the Karup catchment additionally for both the distributed and uniform upscaling procedures. The validation tests described below only considers the 1 km grid model runs, while the remaining results are discussed further below in the section dealing with scaling effects.

### 4.1. Karup catchment

The Karup model (1 km grid) was validated by comparison of model simulations and field data on the following aspects:

- *Annual water balances.* Table 2 shows the annual water balances for the five years simulation period together with the observed annual discharge. The

Table 3
Water balance in mm/year for the Odense catchment at station 45.21 Ejby Mølle (536 km$^2$)

| Year | Precipitation | River flow | | | Observed |
|---|---|---|---|---|---|
| | | Model 1 km grid | Model 2 km grid | Model 4 km grid | |
| 1989 | 649 | 220 | 177 | 187 | 181 |
| 1990 | 943 | 349 | 351 | 394 | 299 |
| 1991 | 760 | 312 | 291 | 308 | 265 |
| 1992 | 770 | 308 | 306 | 332 | 243 |
| 1993 | 906 | 334 | 329 | 353 | 306 |
| Average | 805 | 305 | 291 | 315 | 259 |

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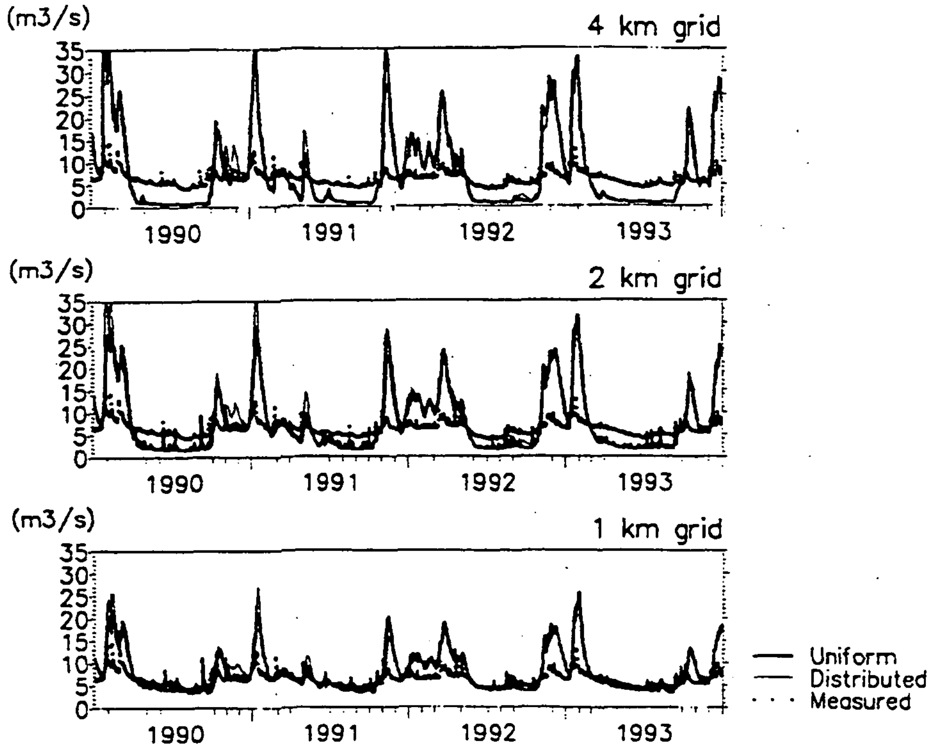



Fig. 5. Comparison of the recorded discharge hydrograph for the Karup catchment with simulations based on 1, 2 and 4 km grids. The two simulated curves corresponds to the combined upscaling/aggregation procedure (Distributed) and the simpler upscaling procedure (Uniform).

simulated and observed hydrographs are shown in Fig. 5.

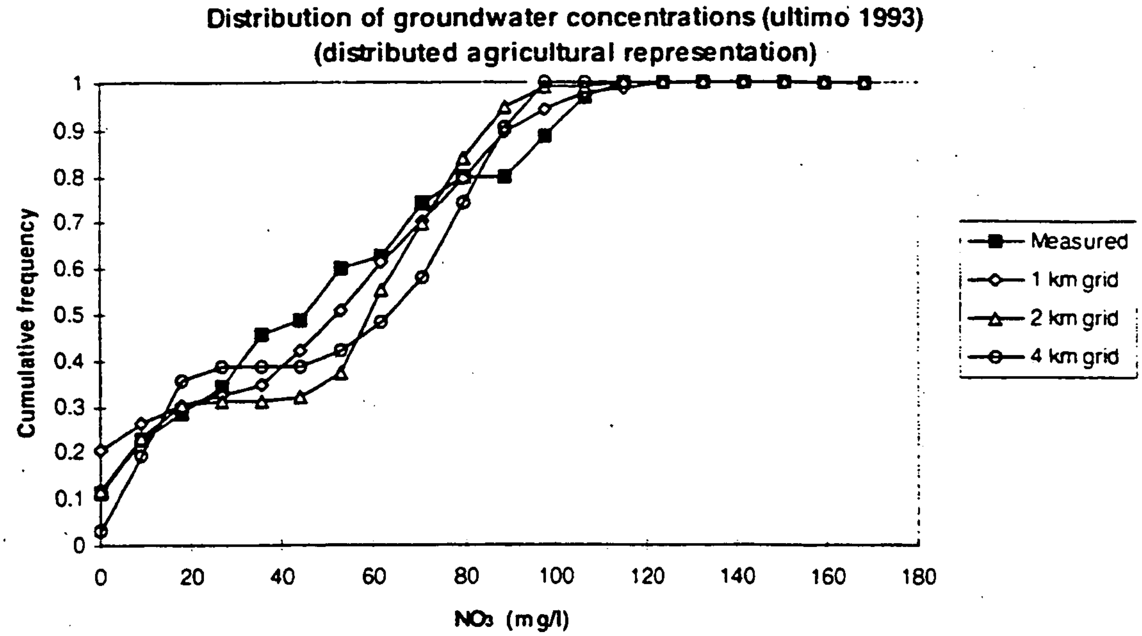
- *Nitrate concentrations in the upper groundwater layer.* Simulated values are compared to observed values from 35 wells in terms of statistical distributions over the aquifer (Fig. 6).

The main findings from these validation tests can be summarised as follows:

- The annual water balance is simulated remarkably well. Thus the simulated and recorded flows, which also reflect the annual groundwater recharges in this area, differ only 2% as average values over the five year simulation period (Table 2).
- The variation of the river runoff over the year is relatively well described, although not at all as good as the long term average water balance (Fig. 6). The model generally underestimates the runoff in the summer periods (low flows) and overestimates the winter flow. There may be many reasons for this. The most important is probably that the observed groundwater levels and dynamics are poorly reproduced by the model. The runoff from the Karup catchment is dominated by drainage flow and baseflow components. Thus a good simulation of groundwater levels and dynamics are required in order to produce a good runoff simulation. An improved simulation of groundwater levels and dynamics requires that the model includes, in particular, spatial variations of the transmissivity of the aquifer, which is not pos. based on the available input data.
- The nitrate concentrations simulated by the model are seen to match the observed data remarkably well, both with respect to average concentrations and statistical distribution of concentrations within the catchment. It may be noticed that the critical $NO_3$ concentration level of 50 mg/l (maximum admissible concentration according to drinking water standards) is exceeded in about 60% of the area.

### 4.2. Odense catchment

The Odense model (1 km grid) was validated by



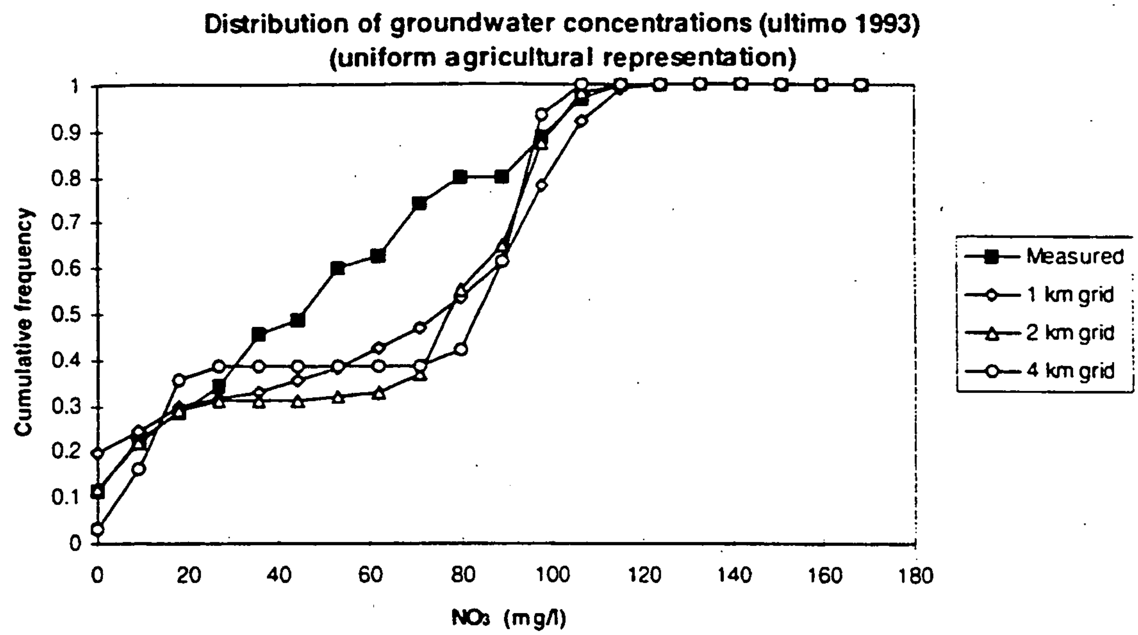




Fig. 6. Comparison of the statistical distribution of nitrate concentrations in groundwater for the Karup catchment predicted by the model with 1, 2 and 4 km grids and observed in 35 wells. The upper figure corresponds to the upscaling/aggregation procedure resulting in a distributed representation of agricultural crops, while the lower figure is from the run with the upscaling procedure, where all the agricultural area is represented by one uniform crop.

comparison of model simulations and field data on the following aspects:

- *Annual water balances.* Table 3 shows the annual water balances for the five years simulation period together with the observed annual discharge. The simulated and observed hydrographs are shown in Fig. 7.
- *Nitrate concentrations in the upper groundwater layer.* Simulated values are compared to observed values from 42 wells in terms of statistical distributions over the aquifer (Fig. 8).

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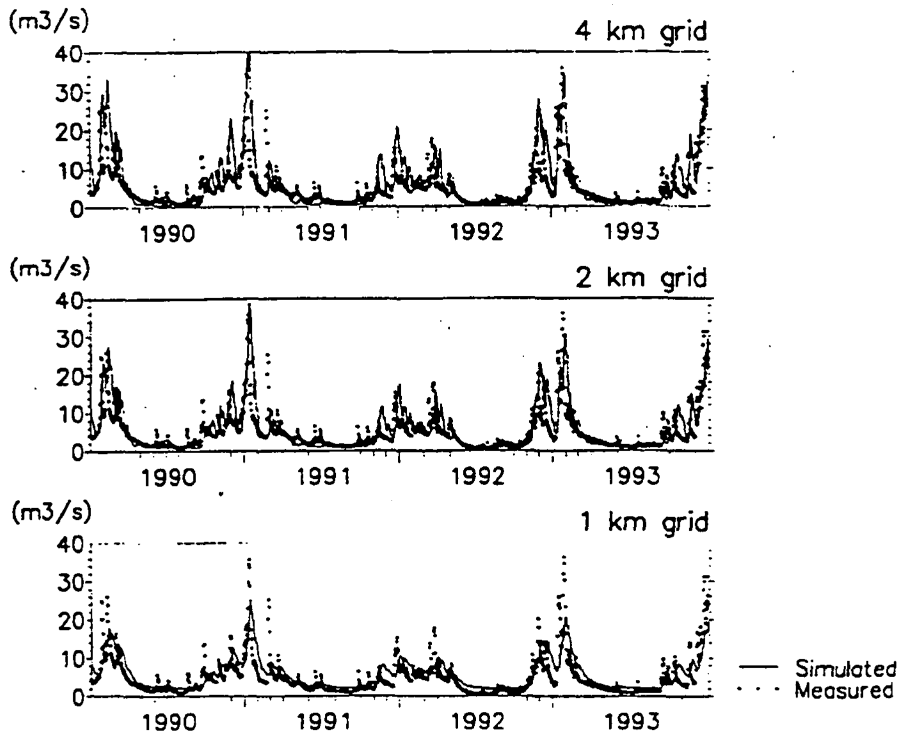


Fig. 7. Discharge hydrographs for Odense catchment simulated with 1, 2 and 4 km grids.

The main findings from these validation tests are:

- The annual water balance is simulated reasonably well, although not with the same accuracy as for the Karup catchment. Thus the simulated and recorded flows differ 18% for the 1 km grid model as average values over the five year simula tion period (Table 3). A comparison with anothe model study for this area reveals that one of th reasons for this deviation is uncertainties (errors) i the catchment delineation in the flat downstrean part of the catchment. Another reason may be tha

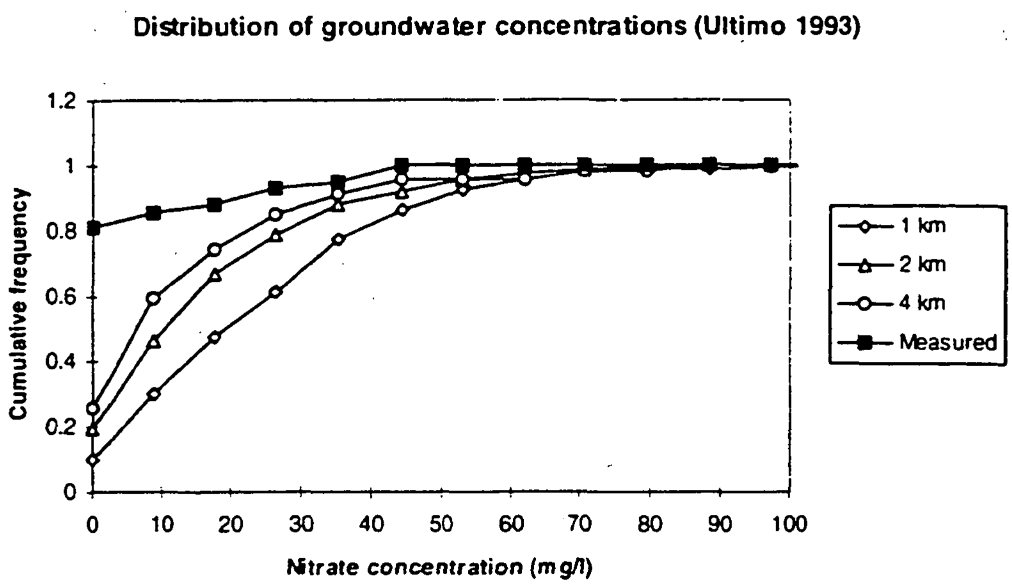


Fig. 8. Comparison of the statistical distribution of nitrate concentrations in groundwater for the Odense catchment predicted by the model witl 1, 2 and 4 km grids and observed in 35 wells.

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the soil hydraulic conductivity functions and the soil water retention curves that significantly affect the evapotranspiration are not very accurately determined. These inaccuracies may originate either from non-representative soil texture data in the 1 km × 1 km GISCO database or by errors introduced by use of the pedo-transfer functions.

- The variation of the river runoff over the year is relatively well described, although the winter peaks are simulated too small and the summer low flows too high, reflecting that some of the internal hydrological processes may not be simulated correctly.
- The distribution of groundwater concentrations by the end of the simulation period is seen not to compare very well to the observations from 42 wells. Thus, in 80% of the observation wells no nitrate was found, whereas the model simulates zero concentration in only 25% of the area. With respect to the critical concentration value of 50 mg/l, the observations indicate that such high concentrations are not found in the area, while the model simulates such concentrations to exist in about 5% of the catchment area. The main reason for this disagreement is most likely that in reality the nitrate is in most of the area reduced (disappears) in the confining sediments overlaying the aquifer. This is not simulated by the model, because the reduction front was assumed to be located within the aquifer, while analysis of local geological data reveals that it in reality is located in the upper confining layer over most of the aquifer.

  It is noticed that the nitrate concentrations are significantly lower in the Odense catchment than in the Karup catchment, both the observed and the simulated values. The main reason for this is that the different soil properties and the less number of animals result in a lower nitrate leaching from the root zone in the Odense catchment.

### 4.3. Scaling effects

The results of running the Karup and Odense models with different computational grid sizes, 1, 2 and 4 km, appear from Tables 2 and 3 for annual water balances and Figs. 5 and 7 for discharge hydrographs. Further, the results in terms of groundwater concentrations are shown in Figs. 6 and 8. From these results the following findings appear:

- The simulated annual runoff is almost identical and thus independent of grid sizes. A reason for some of the small differences is that the catchment areas in the 1, 2 and 4 km models are not quite identical. Thus, the root zone processes responsible for generating the evapotranspiration and consequently the runoff does not appear to be scale dependent as long as the statistical properties of the soil and vegetation types are preserved, which is the case with the upscaling/aggregation procedure used in this case.
- The hydrograph shape differs significantly for the three grid sizes. For the Karup model, the simulation with 1 km grid reproduces the low flow conditions reasonably well, whereas the 2 and 4 km grids have a rather poor description of the baseflow recession in general and the low flow conditions in particular. For the Odense model, the simulation with the 1 km grid shows too large baseflows during the low flow season, while the 2 km grid model has the right level and the 4 km grid model simulates less low flow than observed. This indicates that there are significant scale effects on the stream–aquifer interaction that are not properly described in the present upscaling/aggregation procedure.
- The nitrate concentrations in the groundwater is not clearly influenced by the grid size for the Karup catchment, while there appears to be some effect for the Odense catchment. The reason for this difference is related to the different hydrogeological situations in the two catchments. In the Karup catchment the groundwater table is generally located a couple of meters below terrain surface and the horizontal flows take place in both the Quaternary and the Miocene sediments. Hence for both the 1, 2 and 4 km grid models, the main part of the horizontal groundwater flow takes place in the about 15 m of the aquifer located above the reduction front, and only a relatively small part of the flow lines are crossing the reduction front, below which the nitrate disappears. In the Odense catchment, the horizontal groundwater flows take place almost exclusively in the lower aquifer, of which only the upper 3 m is located

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above the reduction front. This implies that a large part of the groundwater flow is crossing the reduction front on its route from the infiltration zones in the hilly areas towards the discharge zones near the river. As the size of the grid influences the smoothness of the aquifer geometry, the grid size will significantly influence the number of flow lines crossing the reduction front and hence the nitrate concentrations. Such scaling effect on geological conditions is not accounted for in the present upscaling/aggregation procedure.

Further, for evaluating the importance of the combined upscaling/aggregation method ('distributed') a model run has been carried out for the Karup catchment with another upscaling method. This alternative method is based on upscaling of soil/crop types all the way from point scale to catchment scale. This implies that all the agricultural area is described by one representative ('uniform') crop instead of the 17 cropping patterns used in the 'distributed' method. This representative crop has been assumed to have the same characteristics as the dominant crop, namely winter wheat, and further to be fertilised by the same total amount of the organic manure as in the other simulations, supplemented by some mineral fertiliser up to the nitrate amount prescribed in the norms defined by Plantedirektoratet (1996).

The results are illustrated in Figs. 5 and 6 by the legend denoted 'uniform'. The effects on the discharge hydrographs (Fig. 5) are seen to be negligible, indicating that the dominant crop (by chance) has similar evapotranspiration characteristics as the sum of the different crops weighted according to their actual occurrence. The nitrate concentrations in groundwater (Fig. 6) show some differences in terms of a lower average concentration and a less smooth areal distribution as compared to the distributed agricultural representation. Thus, in case of the 'uniform' representation the nitrate concentrations fall in two main groups. Around 30% of the area, corresponding to the natural areas with no nitrate leaching, has concentrations between 0 and 20 mg/l, while the remaining 70%, corresponding to the agricultural area with the 'uniform' crop, has concentrations between 70 and 90 mg/l. In the 'distributed' agricultural representation the areal distribution curve is much smoother in accordance with the measured data.

## 5. Discussion and conclusions

Two prerequisites are required for performing larg scale simulations of nitrate leaching on an operationa basis: firstly access to readily available global (or i the present case European) databases, and secondly a adequate scaling enabling suitable models to b applied at a larger scale than the field scales fo which they usually have been proven valid. A ke challenge as compared to the experiences reporte in the literature is then how to make use of the physi cally based model at large scale without possibility fo detailed calibration at that scale, when we know tha its physically based equations are developed for smal scales. Such model can only be stated as well prove for small scales, and the few attempts made so far t use it on scales above 1000 km$^2$ have applied c tion at that scale (Refsgaard et al. 1998b, 1992; Jain e al., 1992).

### 5.1. Data availability

From the experiences gathered and the lesson learnt with regard to availability of European dat bases the following conclusions can be drawn:

- Not all of the existing "European" databases ar generally applicable due to various restriction (e.g. copyright, not open to other projects, pointer only).
- Not all databases maintained by international insti tutions contain harmonised and integrated dat sets. Many databases in fact only contain a c tion of national data sets that are neither integrate in one seamless data set, nor harmonised in thei contents or nomenclatures.
- Not all input data requirements could be satisfie from GIS (spatial) data sets, why tables and pape maps are needed to supplement the informatior However often the available data are too coars in scale (e.g. EU statistics at a higher administra tive unit than needed) or too specific (e.g. transfe functions for natural soils only but not for agricu tural soils).
- Use of national data sets is to some extent neces sary, with restrictions to data quality and origin.
- The search for data sets could have been largel improved by the existence of a European spati

data clearinghouse and the association of the available data sets with meta information.

It is noted that in spite of comprehensive efforts made during recent years for assessing spatial data by use of advanced remote sensing technology the only data in the "European" databases which originate from remote sensing data are the CORINE land cover data, which were useful for distinguishing between natural, urban and agricultural areas, but which did not contain any further information about agricultural crops of importance in the present context.

In spite of the above limitations, the attempts in the present study to identify suitable data sources at the European scale have shown that useful data are available at that scale for most of the required model input data. Although these data require some kind of transformation, as e.g. pedo-transfer functions, the data appear adequate for overall model simulations at this scale. However, some gaps exist in the European level databases. Thus, for the following data it was necessary to use national data sources:

- Meteorological data on a daily basis.
- Soil organic matter from arable land.
- Agricultural statistics.
- Agricultural practices.

These data were all easily available at a national scale, and hence their availability is not expected to pose significant constraints for large scale modelling in other parts of Europe.

The most critical data that may cause problems in terms of availability at larger scale are the geological data, for which no global (or European) digital database apparently exists. The present case study relied heavily on an EC report produced by the Danish Geological Survey. The information in this report proved adequate for the present purpose, although the lack of geochemical information turned out to have some importance for one of the two catchments. Similar readily available EC reports exist for other countries, but they appear to be non-standardised and comprise information at a variable level of details. Hence, the positive conclusions from using the geological data in EC (1982) for Denmark cannot necessarily be generalised.

### 5.2. *Parameter assessment—no calibration*

An important element of the present methodology is the principle not to carry out any calibration. The parameter values were assessed in three different ways:

- Directly from the available data, e.g. topography and geology.
- Indirectly from the available data through application of predefined transfer functions, e.g. the soil hydraulic parameters.
- Use of standard parameter values that have been assessed in previous studies on other locations.

While the first two methods can be characterised as fully objective and transparent, it may be argued that there always will be some elements of subjective assessment hidden in the use of standard parameter values and that the possible calibration exercises in previous studies may question the "no calibration" statement.

In the present case the standard parameters originate from two model codes and associated accumulated experiences:

- *Parameters in the MIKE SHE part.* The standard parameter used here is the time constant for routing of groundwater to drains (50 days). From comprehensive hydrological modelling experience on dozens of Danish catchments starting with Refsgaard and Hansen (1982) this value can be characterised as a typical value. It is not the optimal value that would be estimated in a calibration for any of the two respective catchments: Thus, for instance the calibrated value for Karup was in Refsgaard (1997) estimated to 33 days.
- *Parameters in the Daisy part.* The standard parameters used here are the ones controlling the vegetation part of the evapotranspiration and the nitrogen turnover processes in the root zone. These parameters are essential both for the water balance and the nitrogen concentrations. The Daisy has standard parameter that can be used if no calibration is possible (or desirable). These standard parameter values have originally been assessed from agricultural field experiments on plot scales (Hansen et al, 1990). As then the process descriptions and associated standard parameter values

have only been subject to minor adjustments through a number of additional tests on new data sets from different countries. It should be emphasised that Daisy has not previously been calibrated on the Karup and Odense catchments. These two catchments, and in particular the Karup catchment, have been subject to modelling studies which have included calibration of the water balance (evapotranspiration) parameters. However, in the previous studies of the Karup catchment (Styczen and Storm, 1993) and (Refsgaard, 1997) the water balance in the root zone was simulated by MIKE SHE, which is not the case in the present study. As the process descriptions for evapotranspiration in MIKE SHE and Daisy are fundamentally different, the Daisy standard parameters used in the present study, have not been affected at all by the previous MIKE SHE studies in the same catchment.

Thus although it may correctly be argued that the standard model parameters are results of previous studies where calibration was carried out, the specific parameters used in the present study have not been subject to, and are not results of, calibration neither in the Karup nor the Odense catchments.

In our opinion, one of the strengths of physically based models is the possibility to assess many parameter values from standard values, achieved from experience through a number of other applications. We think that the results of the present study shows both this strength and some of limitations in this respect. Thus on one hand, the key results in terms of annual runoff and nitrogen concentration distributions are encouraging, while on the contrary Figs. 5 and 7 clearly illustrate that it would be very easy to obtain a better hydrograph fit through calibration of a couple of parameter values.

When parameter values are assessed in this way they inevitably are subject to considerable uncertainty, which again will generate significant uncertainty in model results. It is therefore highly relevant to conduct uncertainty analyses in order to assess whether the resulting uncertainty becomes so large that the model results are not of any use for water management in practise. A methodology and some results of such uncertainty analyses are provided in Hansen et al. (1999) for the root zone processes and in Refsgaard et al. (1998a) for the catchment processes.

### 5.3. Upscaling

The adopted upscaling methodology is a combination of upscaling and aggregation. Hence, upscaling in its traditional definition (Beven, 1995) is used only from point scale to field scale, where the same equations are assumed valid and where 'effective' parameter values are used. The parameter values estimated through pedo-transfer functions (soil data) and the vegetation parameters representing the different crops are assumed valid at field scale. Subsequently, an aggregation procedure is used to represent catchment scale conditions with regard to soil and vegetation types. This aggregation procedure is in full agreement with the findings made regarding the apparent existence of a threshold area (R[illegible]) above which "... spatial patterns of dominant pro[illegible] controls can be represented by their statistical distribution functions" (Famiglietti and Wood, 1995).

This theoretical consideration is supported empirically by the model results, which show that the annual catchment runoff can be simulated well, even when using different model grid sizes. For the Karup catchment, where the nitrate reduction in the aquifer does not appear to have influenced the results adversely, even the statistical distribution of nitrate concentrations is simulated well.

For simulation of annual runoff and nitrate concentration distributions, both of which are affected primarily by root zone processes, the impact of changes of scale is thus relatively small. In contrary to this, the impact on hydrograph shape is consistently rather large. This finding, which also is docume[illegible] earlier in Refsgaard (1997), indicates that the appl[illegible] upscaling/aggregation procedure has important limitations with regard to describing the stream–aquifer interactions. Thus in summary, upscaling of processes described by vertical, non-correlated, but patchy, columns is successful, while the upscaling fails in case of processes where horizontal flows between grids dominate. The differences in hydrograph shapes caused by the differences in grid sizes illustrate how careful a model user has to be when changing grid size. In our opinion it is not relevant to talk about an 'optimal' scale for hydrograph simulation. The important point is rather that the present methodology is scale dependent with regard to hydrograph simulation; hence a change of scale (grid size)

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generates a need for recalibration of parameters responsible for baseflow recession and low flow simulation.

An alternative, and commonly used, upscaling procedure, where upscaling is used all the way from point scale to catchment scale by selecting the dominant crop type in each grid, resulted in one uniform crop representing all the agricultural area. Results indicate that whereas this uniform upscaling procedure may be sufficient for simulating annual water balance and discharge hydrographs, it is not satisfactory for simulation of nitrate leaching and groundwater concentrations. This is in agreement with Beven (1995) who states that upscaling from small scales to larger scales using effective parameter values cannot be assumed to be generally adequate.

An inherent limitation of the applied upscaling/aggregation method is that it does not preserve the georeferenced location of simulated concentrations, but only their statistical distribution over the catchment area. Therefore, comparisons with field data make no sense on a well by well or subcatchment by subcatchment basis, and no information on the actual location of the simulated "hot spots" within the catchment is possible. If it from a management point of view is required with a more detailed spatial resolution of the model predictions, then the same upscaling method has to be carried out at a finer scale with all the statistical input data being supplied on a subcatchment basis. This is in principle straightforward, but in reality it may often be limited by data availability.

A critical assumption in the upscaling procedure is the application of the point scale equations at the field scale with effective parameters. This corresponds to interpreting the field as a single equivalent soil column using effective hydraulic parameters. This approach was evaluated on two Danish experimental 0.25 ha plots, a coarse sandy soil and sandy loam, using the Daisy model (Djurhuus et al., 1999). The two plots were monitored with respect to soil water content and nitrate in soil water at several depths at 57 points, where also texture, soil water retention and hydraulic conductivity functions had been measured. The conclusions from comparing the field measured data with the model simulations over the experimental plot, represented by the 57 points, was that the observed mean nitrate concentrations were matched well by a simulation using the geometric means as effective parameters. This conclusion is in agreement with previous studies for Danish hydrological regime (Jensen and Refsgaard 1991a–c; Jensen and Mantoglou, 1992). Other studies from other regimes (Bresler and Dagan, 1983) conclude that effective soil hydraulic parameters are not adequate for modelling water flow in spatially variable fields. The critical issue determining whether such approach is feasible or not may depend on whether Hortonian overland flow is created in the hydrological regime in question. Thus, although the upscaling methodology from point to field scale is far from universally valid, there are good reasons to believe that this assumption was satisfactorily fulfilled in the present case studies.

The spatial patterns, which in subsurface hydrology is considered to be of significant importance (Wen and Gómez Hernández, 1996), have been treated in different ways with regard to continuous data (parameter values) and categorical data (soil and vegetation classes). The effects of spatial autocorrelation of soil and vegetation parameters within a field have been assumed incorporated into the 'effective parameters', which in the present case are assessed in a rather crude way through pedo-transfer functions and use of standard values. The categorical data have been treated differently in the aggregation procedure for soil and vegetation classes. The soil data (one soil type for Karup and two soil types for Odense) were assessed from the soil map and assigned at a grid basis so that the percentage of each soil type within a catchment was preserved and the individual grids to the largest possible extent were characterised by the dominant soil type within the respective grid. For the vegetation types, the same procedure was applied to initially distinguish between agricultural and non-agricultural areas by use of the land cover map. Subsequently, it was assumed that the spatial distribution of cropping patterns are random and without spatial autocorrelation. This is justified by the agricultural management practise of rotating the crops within the individual farms.

### 5.4. General applicability of methodology

From the results of the present study it appears that it is possible to use distributed physically based models of the same type as the MIKE SHE/Daisy for catchment scale assessment of nitrate contamination from agricultural land. It appears obvious that such model application is straightforward and the above conclusion is valid for other areas in Denmark. The interesting question is therefore how general this conclusion is to other areas in Europe (and on other continents) and what the scientific and practical limitations are. In this respect the following considerations may be noted:

- Except for the geological data, the general availability of which are somewhat uncertain, there is no reason to expect that the application of similar data for other catchments in other European countries should not be as relatively easy as the application for the two Danish catchments. Likewise, the encouraging simulation results of using European level databases, in spite of their often coarse resolution and high level of aggregation, may also be expected for other areas. With regard to geological data it may be noted that considerable efforts are being made at most (if not all) national geological institutes to provide geological data to users in digital form; hence the limitation on non-easy data availability existing so far is likely to be overcome, at least nationally, during the coming years.
- The combined aggregation/upscaling procedure appears valid in many areas. The catchments for which it was used in the present study were limited to a maximum of about 500 km$^2$. However, the further upscaling to larger areas provides no fundamental problems, as it consists of just a larger number of computational grids. Computationally, running a model like MIKE SHE/Daisy for an area of for instance 100 000 km$^2$ with e.g. 250 subcatchments of each 100 grids is maybe close to the limit of what is practically feasible today (five years run would require 100 h CPU time on a Pentium 300 MHz), but this problem will soon disappear as computers become faster.
- The MIKE SHE/Daisy modelling methodology is general and applicable to many other areas. Some limitations, however, is related to special geological conditions such as karstic flow and fissured aquifers, which cannot be described explicitly. Another important limitation is related to the upscaling procedure from point to field scale, which may fail in areas where Hortonian overland flow is a dominant mechanism. In this respect it should be noted that many areas with dominant overland flow regimes are mountainous regions characterised by thin soil layers and steep slopes, which generally not are regions with important aquifers.

Hence, it may be concluded that the methodology can relatively easily be applied to larger areas and used as decision support tool for evaluation of legislative and management measures aiming at reducing nitrate contamination risks.

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# Application of irrigation optimisation system (IOS) to a major irrigation project in India

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**Abstract.** The Irrigation Optimisation System (IOS), a decision support tool and an extension of the combined hydraulic-hydrological modelling system for canal-command, is applied as a planning tool to the Mahanadi Reservoir Irrigation Scheme, a large irrigation project in Central India. Besides the MIKE 11 and MIKE SHE modelling systems, for the hydraulic and hydrological simulations respectively, it has an optimisation module to govern the canal releases. The results show that presently the canal releases are inefficient and lead to the wastage of a significant amount of water during the monsoon season. It is shown that the application of IOS reduces this wastage and results in higher irrigation intensity and physical productivity of water in the command. The case study also illustrates the capability of IOS as a planning and decision support tool.

**Key words:** India, irrigation, optimisation, scheduling

## Introduction

The operation and management of irrigation systems are of growing concern worldwide. This is especially true for developing countries, particularly in Asia, where the need to enhance the agricultural productivity is coupled with decreased availability of water for agriculture owing to rapid industrialisation and ever-increasing municipal needs (Biswas 1994; Lenton 1994). The irrigation systems in the region are predominantly operated by manual upstream control. Further, the lack of knowledge about irrigation and water management among farmers makes the management of the whole scheme the primary responsibility of the irrigation departments (districts/agencies). Since these departments presently do not possess any scientifically based decision support system, the water allocations are frequently subject to negotiations with the farmers and politicians (Wade 1983). This results in the ignorance of crop water demands with respect to time and quantity of water and leads to the poor performance of irrigation projects (ASCE Task Committee 1993).

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In recent years, several canal automation and control algorithms have been developed (Clemmens & Replogle 1989; Loof et al. 1991a; Malaterre. 1995). These advanced technologies, aimed at improved irrigation system management, are presently not feasible for on-line control in the vast majority of the existing irrigation systems in Asia due to lack of financial resources and infrastructure. However, these can be implemented for supervising manual controls and providing canal control guidelines.

Due to increased stress on improved irrigation management and planning, a number of optimisation and simulation models have also been developed by various researchers (Dudley et al. 1971; Lakshminarayana & Rajgopalan 1977; Yaron & Harpinist 1980; Yaron et al. 1987; Paudyal & Das Gupta 1990; Loof et al. 1991b; Onta et al. 1991; Chavez-Morales et al. 1992; Srivastava & Patel 1992; Steiner & Keller 1992; Burton 1994; Loof et al. 1994; Onta et al. 1995). However, most of these models are site-specific and address local problems. Moreover, these models focus narrowly on either hydraulic or hydrological aspects of the irrigation system and, therefore, may not meet the objectives of the irrigation departments dealing with the whole scheme.

The Irrigation Optimisation System (IOS) developed at the Danish Hydraulic Institute and presented here is an attempt to provide a decision support tool with which irrigation agencies can optimise the canal releases to meet the crop water demands within existing infrastructure. IOS is an extension of the modelling system that combines the hydraulic simulations of the canal and hydrological simulations of the irrigated command, using MIKE 11 and MIKE SHE, two well-established modelling systems (Singh et al. 1997). In the existing modelling system an optimisation module is added for governing the canal releases. IOS also has a user-interface that offers operational ease and makes it easily adaptable. IOS is applied to a large irrigation project in India to illustrate its advantages in improved irrigation system management.

## Description of irrigation optimisation system (IO)

The main objective of introducing and implementing a computer-based decision support tool in an irrigation system is to improve the system performance for crop yield, water use efficiency, environmental sustainability or any other criterion decided by the management. The means of controlling the system depends on the time horizon in the operation. In long-term planning, the active means are typically the land use patterns together with the assessment of releases from the major reservoir(s). Since hydrometeorological conditions cannot be forecasted deterministically with sufficient accuracy for longer lead times, a stochastic element is often included here. On the other hand, short-term operation, up to a maximum of two weeks, can be treated

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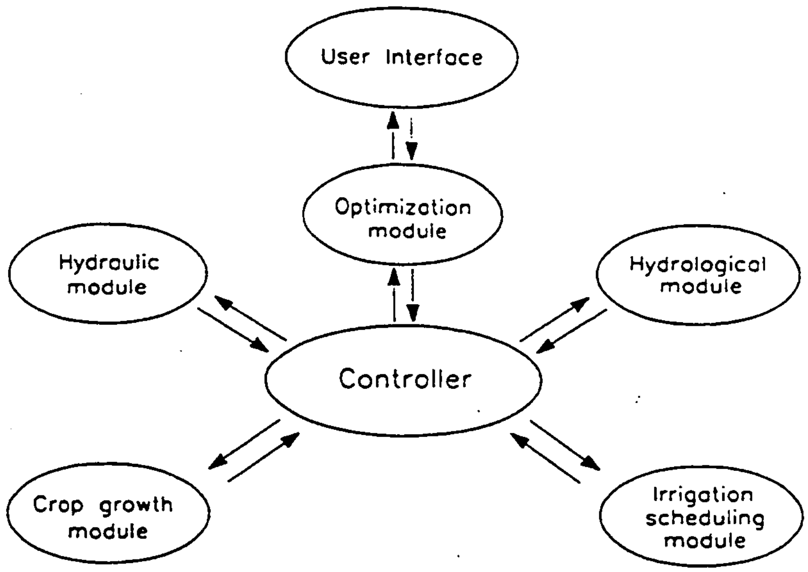


*Figure 1.* Irrigation Optimisation System (IOS).

as deterministic. In short-term operation, the primary means of control are adjustable gates that may be operated to meet the crop water demand in the system. The short-term operation may be used either as a planning tool or for on-line automatic control. In the former, farmers and irrigation managers choose an irrigation schedule for the forthcoming week(s), whereas in the later case real-time data acquisition and efficient optimisation algorithm become crucial. Here, IOS is proposed as a short-term planning tool with decision time steps of two weeks, a realistic time step in most large South Asian irrigation commands.

Figure 1 presents the modular structure of IOS. The core of the system is the *controller module*, which controls and steers data flow among various modules. The details of the optimisation module along with a brief description of other modules are given below. For a more detailed description of hydraulic, hydrological, irrigation scheduling and crop growth modules, see Singh et al., 1997.

### *Hydraulic module*

The transport of water through the canal system is modelled using the *hydrodynamic module* (HD) of the MIKE 11, the one-dimensional unsteady river simulation modelling system (Havnø et al. 1995). It includes the description of flow over a variety of hydraulic structures, generally encountered in an irrigation system. It also includes the possibility of simulating the operation of gates or regulators in canals.

Three options are available for controlling gates and regulators. In option one, the user can specify the regulator position as a function of time.

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In option two, the optimisation module can control the regulators based on the hydrodynamic head in the canal. In the third option, regulators can be controlled on demand by the irrigation scheduling module. However, this option is applicable for off-take regulators only.

*Hydrological module*

The water movement in the irrigated command is modelled by *MIKE SHE*, a distributed physically based modelling system (Abbott et al. 1986a, b; Refsgaard & Storm 1995). MIKE SHE describes the entire land phase of the hydrological cycle in a given command. The model area is discretized by two analogous horizontal-grid square networks for surface and ground water flow components. A finite difference solution of the partial differential equations, describing the processes of overland and channel flow, unsaturated and saturated flow, interception and evapotranspiration, is used for water movement modelling. However, in IOS application, the channel flow routing is performed by MIKE 11, with MIKE SHE coupling for each time step.

*Irrigation scheduling module*

The irrigation scheduling module is based on the water balance technique and uses either the soil moisture approach or the water level approach. In the soil moisture approach, the irrigation demand is governed by user specified maximum allowable depletion (MAD). In the water level approach, used exclusively for paddy (except during ripening and grain formation stages), irrigation demand is governed by the water levels, defined as a function of crop growth stage. Here MIKE SHE performs the water balance calculations, based on which the irrigation scheduling module calculates the irrigation demand.

*Crop growth module*

Modelling of crop growth and crop yield is required for assessing the effect of water stress on crop production. Daily potential and actual yields, leaf area index and yield loss due to moisture stress are the main outputs from this module. The yield loss due to water stress follows the FAO relationship (Doorenbos & Kassam 1979).

*Optimisation module*

The optimisation module employs the deterministic hydraulic, hydrological and crop growth modules, embedded into a non-linear optimisation framework, for the gradient based search leading to improved irrigation system

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operation. This idea, though computationally demanding, offers certain advantages. In large irrigation systems, complex non-linearity associated with the canal hydrodynamics and crop growth plays a major role while describing the system responses to alternate flow schedules. The deterministic models encapsulate the solution to this non-linearity and allow the system analyst to concentrate on the formulation of the objective function.

In an irrigation system, the operational objective is to optimally use the available water for maximising the crop production. To capture this basic logic, a specific objective function is devised in IOS. Here, the objective function is introduced through the evaluation of hydrodynamic states at certain locations in the canal system and crop states on individual fields.

For the hydrodynamic condition, convex non-linear functional relationships of relevant system variables and penalties at certain locations in the system are established. These relationships are typically constructed so that the departure from the 'ideal' system state, i.e., full supply level in the main canal, results in increased penalty. On the other hand, the evaluation of crop yields on individual fields is based on the results from the crop growth module. Here, deviations from the potential crop yield due to water stress results in penalty. These penalties are also filtered through a convex non-linear relationship.

The overall objective function includes these two non-linear penalty functions (hydrodynamic states and crop states) and is defined within the solution space determined by the discrete control variables, i.e., regulator releases at specific time instants. The quasi-continuity in the solution space is achieved by the application of linearly interpolated releases between two successive decisions. Though the objective function evaluation here includes summing up of the evaluations of two different system variables, it is achieved by appropriate scaling of the individual penalty relationships (Yde 1995).

The irrigation system operation is optimised over a certain optimisation interval. The penalties associated with the variables are evaluated at discrete times within the optimisation interval, typically corresponding to the simulation time steps. Subsequently, such 'filtered' time series is integrated over the total optimisation period, T. The total cost (penalty) is, then, defined as

$$C = \int_0^T \left( \sum_{i=1}^{l} Ch_i + \sum_{o=1}^{p} Cf_o \right) dt \qquad (1)$$

where $Ch_i$ = penalty related to hydrodynamic state, l being the number of evaluation points; and $Cf_o$ = penalty related to field, p being the number of fields.

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The reservoir releases, the releases from the head regulators or through the cross- or diversion regulators are optionally defined as the decision variables (control variables) in the optimisation problem. The number of regulators included in the optimisation problem should be kept within reasonable limits to minimise the computational load. The remainder of the regulators may be controlled 'on demand.' The releases from such regulators are maintained within feasible limits by the hydrodynamic model.

The objective function can be mathematically expressed as

$$\text{Minimize } C(q_k^j)\, j = 1, \ldots, m; k = l, \ldots, n \tag{2}$$

where $(C(q_k^k)$ = hydrodynamic and field penalties, integrated over the optimisation interval; $q_k^j$ = flow through control structure k at time j; m = number of time intervals (decision points); and n = number of control structures.

The optimisation module is based on the customised steepest descent method with an open interval Fibonacci line search algorithm. The length of the first search step, corresponding to the final interval of uncertainty, is chosen such that it brings meaningful change in regulator releases. Each following step is increased according to the Fibonacci proportions, until the search interval closes. The detailed descriptions of the optimisation routine and the solution technique are given in Tomicic (1989) and Yde (1995).

*Coupling of modules in IOS*

The different modules in IOS operate interactively. For example, the hydraulic module receives the information on the control of regulators from the optimisation module via the controller while it sends the information about the hydrodynamic state of the canal system. The controller then transmits it to the hydrological module. The hydrological module, based on this information, performs the water balance calculations. It then sends the state of the individual fields in terms of potential and actual evapotranspiration, effective and maximum moisture content integrated over the root depth, recharge to the ground water and depth of the ground water table to the controller. The controller module then transmits this information to the irrigation scheduling and crop growth modules. Based on the information received, the irrigation scheduling module calculates the irrigation demand and sends it to the controller module for onward transmission to the optimisation module, whereas the crop growth module calculates the crop yield, and so on.

It may, however, be noted that the use of optimisation module is optional here. The user may opt for rotational or on-demand irrigation with pre-specified releases.

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## Case study

The case study area is same as in the hydraulic-hydrological modelling study (Singh et al. 1997). IOS is applied to the Mahanadi Reservoir Irrigation Scheme (MRP), consisting of six interlinked reservoirs. It is designed to irrigate an area of 374, 000 ha in kharif (monsoon) season and 131. 000 ha in rabi (winter) season, through five interlinked canal systems, besides meeting the municipal and industrial demands in the adjoining area. However, in the present study, only the Mahanadi main canal command, which accounts for 197, 460 ha of the design area, is used for modelling. Data on various aspects of the command area viz. topography, geology, soils, crops, main canal system and daily canal releases for 1991 are obtained from the Irrigation Department, MRP.

The Mahanadi main canal, with a design discharge capacity of 391 $m^3/s$ at the head end, supplies water to four branch canals, several distributaries and field outlets. The main and branch canals are lined. The main canal, branch canals and distributaries are each equipped with a head regulator. In addition four cross regulators are provided along the main canal. Figure 2 presents the daily releases to the main canal during 1991. As evident from the figure the irrigation releases are limited to only kharif season, a regular feature in the command.

### *Model setup*

The model setup for hydraulic, hydrological, irrigation scheduling and crop growth modules is similar to that used in the hydraulic-hydrological modelling study (Singh et al. 1997). A brief description of these and a detailed description for the optimisation module are presented here.

### *Hydraulic and hydrological modules*

Here the entire irrigated command is merged to represent 64 fields (Figure 3) and only the main canal is considered for the unsteady flow modelling. Consequently, the branch canals and distributaries are merged to 16 distribution channels, each linked to four fields.

Though a lumped approach is used in setting up the hydraulic-hydrological modules, the proportional distribution of soil types and crops are maintained as per MRP records. For this, while merging the branch canals, distributaries and field outlets present in the canal system to represent 16 distribution channels, their command areas are also merged to represent 64 fields, i.e., each distribution channel is linked to four fields, two each for heavy and light soils. This results in the field sizes varying from 90.5 ha to

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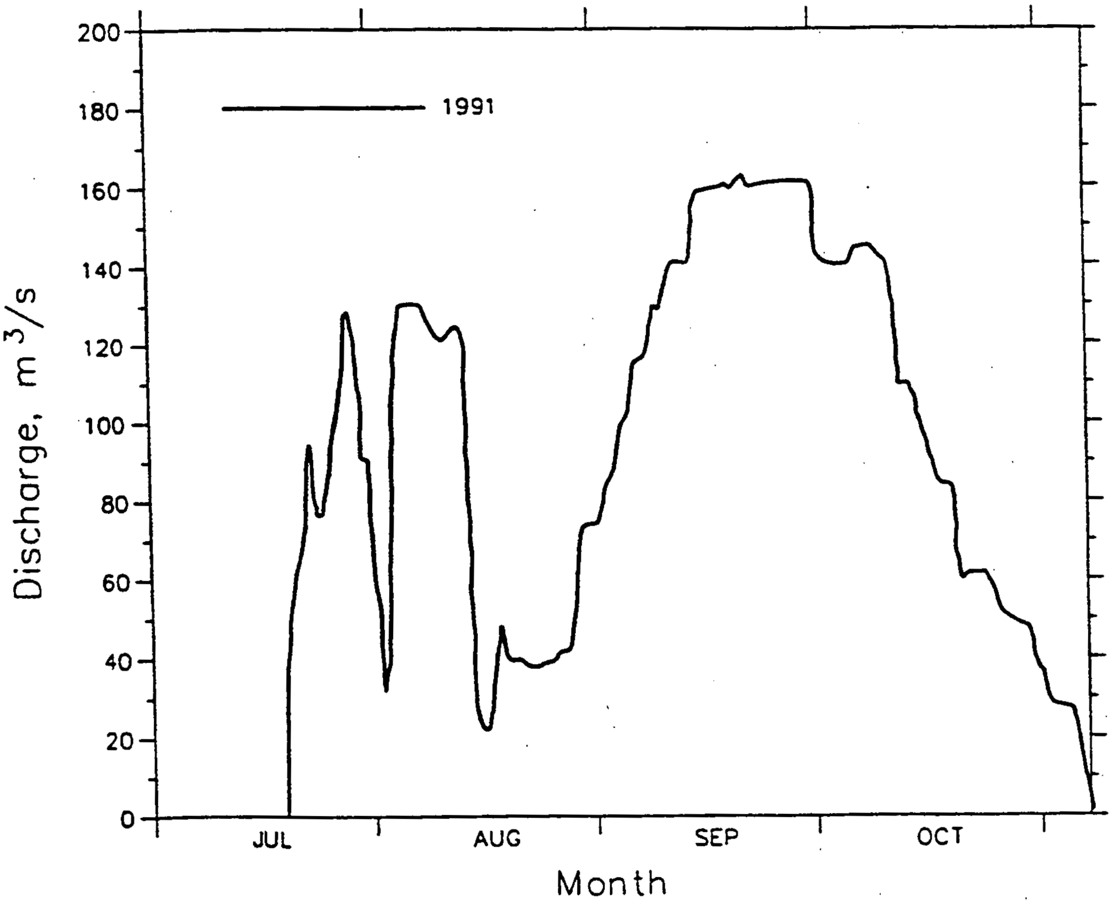


*Figure 2.* Mahanadi main canal releases during 1991.

9344.0 ha. This approach helps in keeping MIKE SHE simulations close to the real field conditions where a large variability in the field sizes are often found. MIKE 11 simulations, however, remain unaffected by the field size variations.

Daily rainfall and pan evaporation data for 1991–92 are taken from an earlier study (Singh et al. 1997). Data on soil physical characteristics and saturated hydraulic conductivity are determined from the literature (Katre 1992; Agrawal 1994). The crops considered here are paddy, transplanted and direct sown in kharif season and wheat, mustard, grain and potato in rabi season. Data on areal distribution, leaf area index (LAI) and maximum root depth of crops are obtained from the Irrigation Department, MRP and Agronomy Department, Indira Gandhi Agricultural University, Raipur. Crop coefficients, $K_c$, for Indian conditions, and yield response factor, $K_y$, are determined from the literature (Mazumdar 1983; Doorenbos & Kassam 1979).

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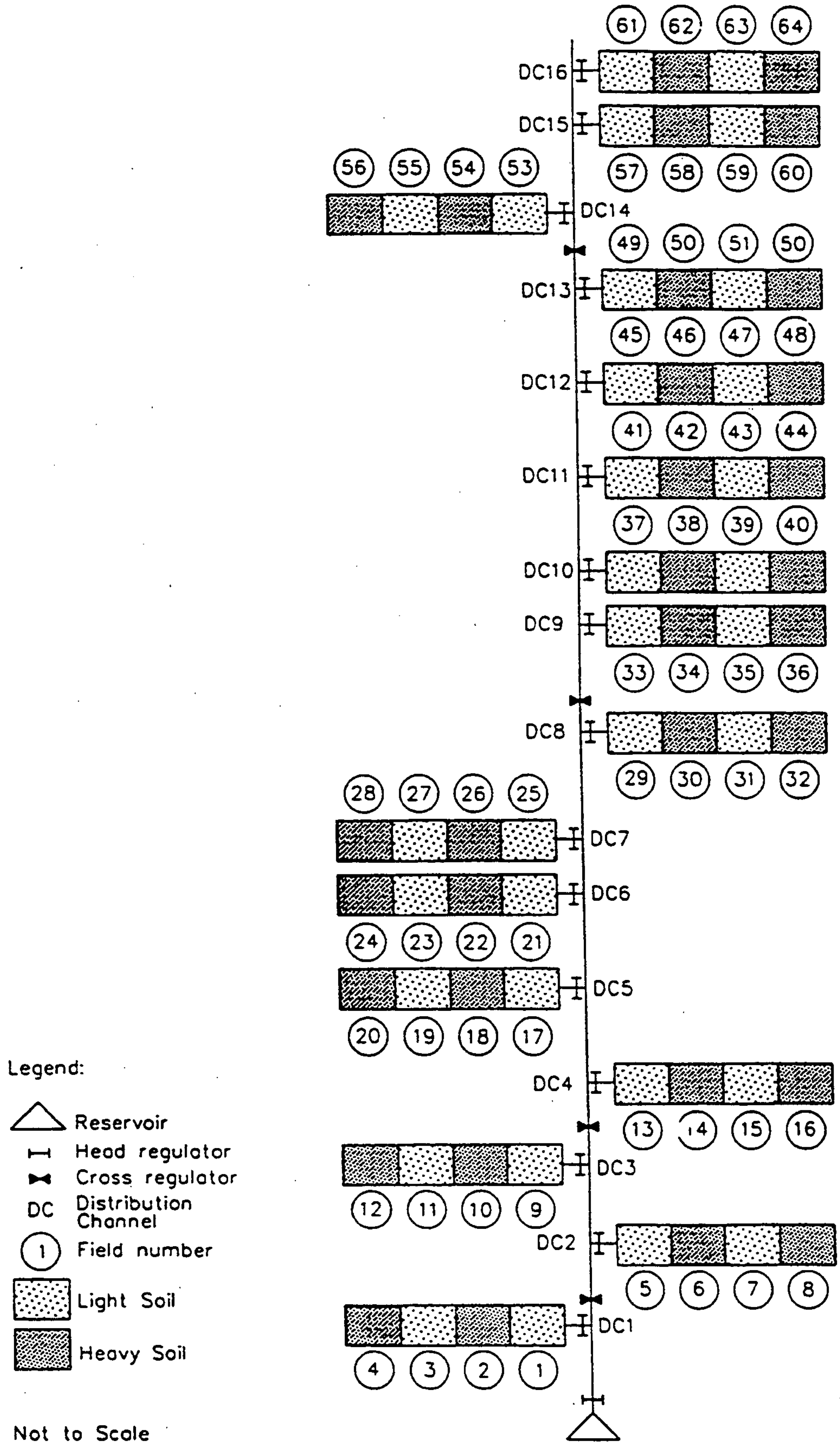


*Figure 3.* Schematisation of the hydraulic-hydrological modules for the Mahanadi Reservoir Irrigation Scheme.

*Irrigation scheduling and crop growth modules*

In the soil moisture approach of irrigation scheduling, a MAD value of 0.50 is used (Stegman 1983). In the water level approach, water levels recommended at different growth stages of paddy are used (Doorenbos & Kassam 1979). Here, the user may also opt for full irrigation or deficit irrigation. In the case study, however, the practice of full irrigation is adopted. In the crop growth module, the maximum LAI, time to maximum LAI and maximum root depth are used to fit the crop growth functions.

*Optimisation module*

Here, only the head regulator of the main canal is considered as the control structure. This is because the canal releases in MRP are presently regulated by this structure only. The cropping season is considered here as the optimisation horizon with simulation time step of one week (the period between the decision points; a time step of one hour is used in the hydraulic and hydrological modules). Though ideally a smaller time step should be used, a larger time step is used here to account for the fact that the head regulator is manually operated. The maximum number of iterations is kept at 20, with 10 line searches in each iteration. In most cases this is found to be sufficient to reach the optimum and in all cases this reduced the penalty considerably.

To define the cost of water to be used in Equation (1), production of paddy is considered as the reference variable and all costs are converted accordingly. It is assumed that 1 $m^3$ of water is required for producing 1 kg of paddy (Doorenbos & Kassam 1979). The cost of water for the individual fields is estimated here based on yield loss calculations in the crop growth module. For calculating the cost of water at the tail end of the system, a cost function is devised (Figure 4). The cost function is defined after several simulations and tests of the optimisation procedure. As shown in the figure, the cost function not only depends on the actual cost of water but also takes into the account the social factor, i.e., it is modified to facilitate the supply of water to all fields irrespective of their location. To ascertain this, a simulation is made and it is seen that a minimum flow of 30 $m^3s^{-1}$ is required at the tail end to maintain sufficient head of water in the main canal to supply the downstream distribution channels. Hence, this value is selected as the optimal point in the cost function. The cost function is further provided with a steep slope on the right hand side to improve the efficiency of the line search algorithm. Further, while using the optimisation module, the total volume of water released in 1991 (Figure 2) is specified as the maximum amount of water that can be released through the head regulator.

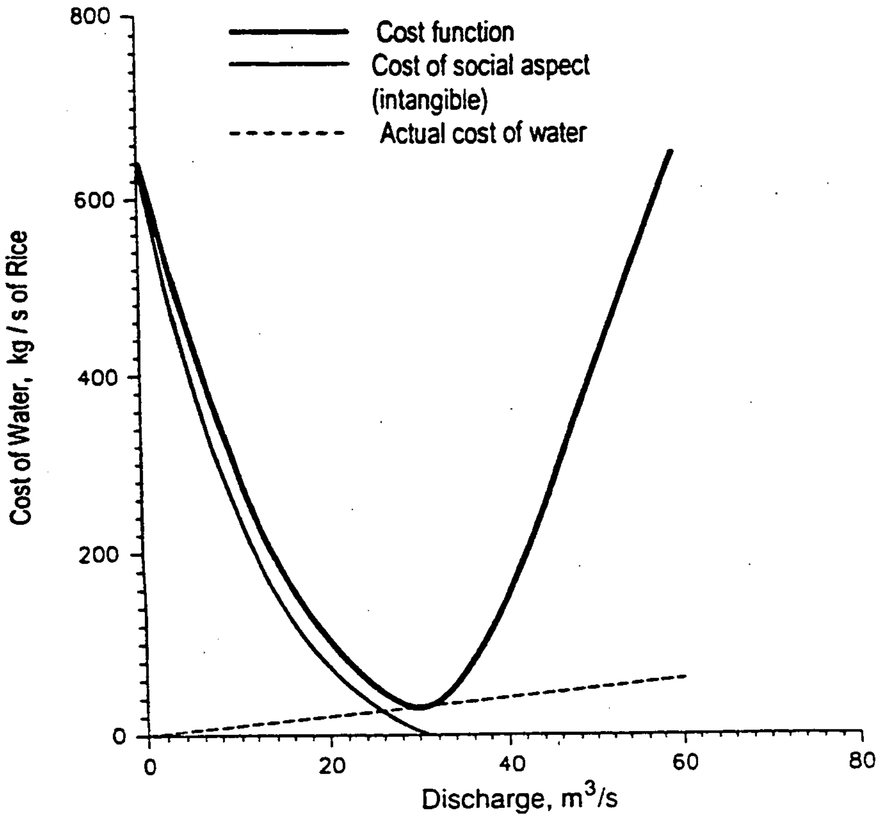


*Figure 4.* Cost function for the water lost at the tail end of the system.

## *Model simulations*

Simulations are made for the 1991 kharif season over a period of fourteen weeks (July 20–October 26). Here the entire command is considered for cultivation, with transplanted and direct sown paddies occupying 28.2% and 71.8% of the area respectively.

Though in practice weekly rotational schedule of distribution channels is adopted in the command area (discussed as Simulation Case I in Singh et al. 1997), here on-demand irrigation, subject to hydrodynamic state of canal system, is used with the actual canal releases determined by the optimisation module.

The calculations are made on a HP 9000/720 workstation with typical computational requirements of 48 hours for optimisation of one full season. For comparison, the speed of the workstation roughly corresponds to that of a Pentium PC with 150 MHz and 32 MB RAM.

To study the impact of the optimisation routine on canal releases and crop production, the simulation results obtained here (referred as "OPTISIM" are summarised in Table 1 along with those obtained in Case I simulation of 1991 of Singh et al. 1997 (referred as "ROTSIM").

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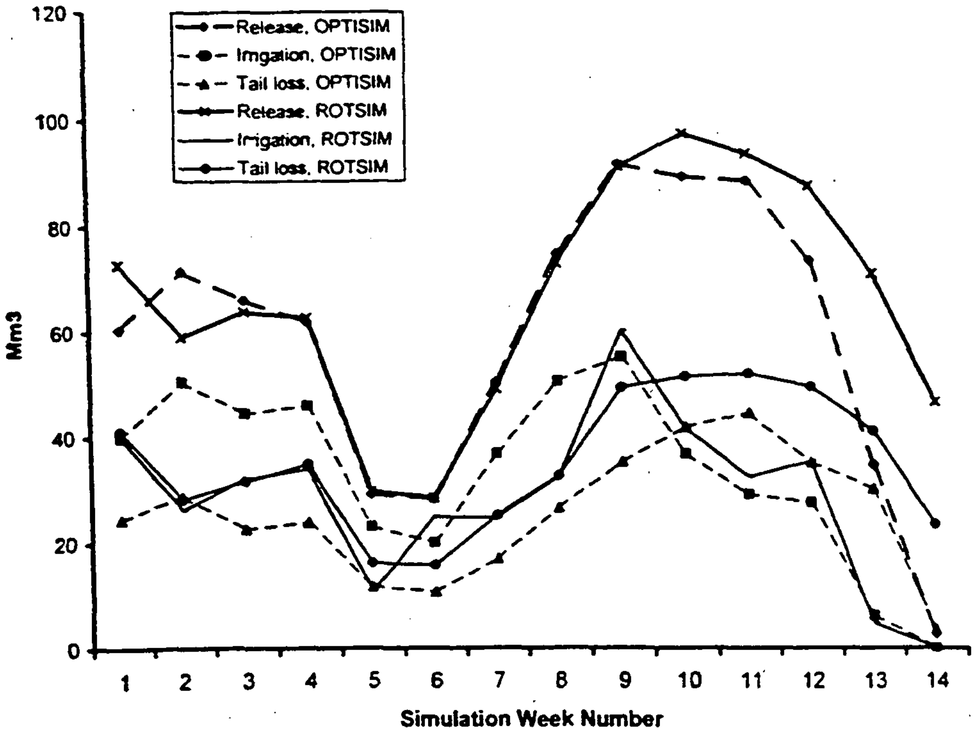


*Figure 5.* Weekly canal releases, tail end losses and irrigation water utilisation.

It is seen that the actual productions are above 98% of the potential production in both cases (Table 1). This shows the adequacy (surplus) of water in the command during the kharif season. Therefore, the optimisation routine focuses on water saving rather than increasing the crop production. Consequently, optimisation routine results in a saving of 112 Mm$^3$ (or 11%) of water compared to the present canal releases. The amount of water saved here is in tune with that reported in Singh et al. 1997, i.e., 120 Mm$^3$, based on weekly canal release and irrigation requirements analysis. It is seen that the optimisation routine in OPTISIM reduces the tail end loss to 43% of the optimised releases (or 38% of the present releases) compared to 53% of the canal releases in ROTSIM. Moreover, it also results in higher irrigation, which is reflected by marginal increase in actual evapotranspiration and actual production. Figure 5 presents the canal releases, irrigation water utilisation and tail end losses for OPTISIM and ROTSIM on a weekly basis. The tail end loss in OPTISIM may further be reduced by considering the cross regulators in the main canal as decision variables in the optimisation problem. However, this is not possible with the actual conditions in the field and would be a much more complex optimisation problem that has not been attempted here.

This shows that the management decisions on fixing the canal releases are presently inefficient resulting in tail losses close to or, at times, even higher than the irrigation water requirement. This excessive wastage explains the

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*Table 1.* Summary of simulation results for the kharif (monsoon) season.

| Simulation case | Potential production | Actual production | Rainfall[3] | Actual evapotranspiration[3] | Irrigation[3] | Canal release | Tail loss |
|---|---|---|---|---|---|---|---|
| | million tons | million tons | mm | mm | mm | $Mm^3$ | $Mm^3$ |
| OPTISIM[1] | 0.621 | 0.614 (98.9%)[4] | 766 | 358 | 235 | 814 | 355 (43%)[5] |
| ROTSIM[2] | 0.621 | 0.612 (98.6%) | 766 | 355 | 215 | 926 | 491 (53%) |

[1]On-demand irrigation with optimised releases
[2]Traditional weekly rotational schedule with actual canal releases (From Singh et al. 1997)
[3]Average values over the entire command
[4]Values in parentheses show % of potential production
[5]Values in parentheses show % of canal release

inability of the management to supply water during the subsequent dry rabi season in the command.

To illustrate the importance of water saving in the kharif season, the OPTISIM of twelve weeks duration is carried out for the rabi season (December-February) with a maximum possible canal release of 112 $Mm^3$. The model setup used here is similar to the one reported for the rabi season in Singh et al. (1997).

To assess the performance of the optimisation routine under short supply conditions, the simulation results obtained here (referred as OPTIRABI) are summarised in Table 2 along with those obtained in a similar rabi simulations of Singh et al. 1997 (referred as "RABISIM").

It is seen that the actual production in OPTIRABI is 88% of the potential production compared to 77% in RABISIM. This is because optimisation routine results in higher irrigation and actual evapotranspiration. The optimisation routine reduces the tail loss from 40% in RABISIM to 23% in OPTIRABI, though the available canal supply is 8 $Mm^3$ less. This saving increases the irrigation and actual evapotranspiration from 101 mm and 187 mm in RABISIM to 135 mm and 193 mm in OPTIRABI respectively. This shows the effectiveness of the optimisation routine in handling the conditions of water scarcity.

Further, the results suggest that it is possible to increase the irrigation intensity from the present 100% (ROTSIM) to 135% (OPTISIM and sav-

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*Table* 2. Summary of rabi simulation results.

| Simulation case | Potential production | Actual production | Rainfall[3] | Actual evapotranspiration[3] | Irrigation[3] | Canal release | Tail loss |
|---|---|---|---|---|---|---|---|
| | million tons | million tons | mm | mm | mm | $Mm^3$ | $Mm^3$ |
| OPTIRABI[1] | 0.143 | 0.126 (88%)[4] | 4 | 193 | 135 | 112 | 26 (23%)[5] |
| RABISIM[2] | 0.143 | 0.110 (77%) | 4 | 187 | 101 | 120 | 48 (40%) |

[1]On-demand irrigation with optimised releases
[2]On-demand irrigation with pre-specified canal releases (From Singh et al. 1997)
[3]Average values over the entire command
[4]Values in parentheses show % of potential production
[5]Values in parentheses show % of canal release

ing of water enabling cropping in the second season over 35% of the area). Correspondingly, the physical productivity of water, an irrigation system performance indicator defined as the ratio of physical quantity of crop production to the volume of water used (Small & Rimal 1996), can be enhanced from the present 0.66 kg $m^{-3}$ (612 Mkg/926 $Mm^3$, ROTSIM, Table 1) to 0.80 kg $m^{-3}$ [(614 + 126) Mkg/(814 + 112) $Mm^3$, OPTISIM and OPTIRABI, Tables 1–2].

## Discussion

### *IOS methodology*

When applying the IOS, the optimisation interval and the time resolution in the decision space have to be harmonised with the type of application and the actual time scale of the processes involved. The nature of the real-time scheduling application imposes a relatively short optimisation interval, which inevitably leads to a 'short-sight' effect. Such an optimisation interval does not cover the effects of the actual release scheduling, which in principle is felt over the entire vegetation season. In practice, the scheduling procedure, consisting of overlapping successive search runs (e.g., the process with one week optimisation horizon and one day time resolution may be repeated daily), allows that the effects due to short-sight and the errors associated with the deterministic forecasts of stochastic boundary variables, are partially com-

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pensated for. Under such application conditions, considering the reasoning behind the specification of the objective function, as well as the implemented relaxed criteria in the optimisation procedure, IOS does not perform the optimisation in its strict terms, but successive searches for an improved system operation.

Establishment of the cost function is crucial. In IOS, the cost function contains two elements, one associated with the water lost at the tail end of the system and the other associated with the loss of crop yield due to water shortage in the fields. The penalties for the water lost at the tail end of the system (Figure 4) are introduced to ensure availability of water in the downstream distribution channels. This may be seen as a social objective benefiting the farmers and societies in the downstream area. The second element in the cost function, which is aiming at increasing the crop yield over the entire area, is a pure economic objective. Thus, the cost function may be considered as a way of combining the two objectives. If the social costs were not considered the result of the optimisation would be full water supply to the fields in the upstream areas and virtually no irrigation in the downstream fields. If the adjustment of gates/cross-regulators in the main canal were feasible, the cost of water lost at the tail end would have to be translated into a cost related to the water level in the main canal.

In the planning applications, with the optimisation horizon determined typically by the vegetation season, the decision resolution may be taken much more roughly, e.g., weekly intervals. In this respect two critical issues on the IOS approach may be raised.

Firstly, it may be argued that a fully dynamic hydraulic model is not needed under such conditions, and that a simpler model could be used instead. This is, seen from a pure optimisation and computational point of view, correct in cases where the travel time is significantly less than the distance between the decision points. However, in other cases with a finer optimisation resolution, a correct description of the canal system dynamics may be important. A key benefit of the present IOS is its generic formulation enabling it to be technically suitable for a wide range of irrigation systems. The main cost of such a generalised system is that it becomes computationally heavy for certain applications. As the aim of IOS has been to develop a generic tool, the higher computational requirement is a 'calculated cost.'

Secondly, more concern in the planning applications should be devoted to the fact that the optimisation is performed under the assumption of perfect knowledge of boundary conditions throughout the season. This is, of course, only possible for the historical records or some synthetic series, derived from statistical properties of long-term historical records. The value of such results

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is related to the identification of optimal scheduling patterns for typical or probable hydrological conditions.

In both cases, the computational time sets the limits for the feasibility of the IOS approach. However, this feasibility limit will, as the price/performance ratio in the computer industry continually decreases, gradually disappear during the coming years. Even today it is computationally feasible to run IOS, with case studies similar to the one presented above, on high performance PCs.

*Case study results*

The travel time in the canals of the present case is about a few days, implying that a fully dynamic hydraulic module might not have been required for canal routing in this case. On the other hand, the memory in the soil moisture system in the fields, determining the irrigation requirements and the crop yields is more than a couple of weeks, indicating that the hydrological module may be important in this case.

The results of this case study suggest that the use and application of IOS as a decision support tool can lead to considerable improvements in irrigation systems like MRP. IOS can help irrigation managers to determine: (1) the canal releases to match the crop water demands in the wet season; (2) how to prepare the rotational schedules of the distributaries to rationalise the water use; and (3) how to use the saved water in the subsequent dry season to enhance the crop production. Thus, this tool can help in planning and operation of canal irrigation systems so that the physical productivity of water can be maximised.

However, since a coarse spatial and temporal discretization is used to describe the model setup in the case study, the results cannot be judged from an absolute viewpoint. Moreover, though IOS, in its present form, is able to analyse the physical aspects of the irrigation system management, socio-economic aspects also need to be taken into account in the irrigation management process. Though the cost function devised and used in this study takes the social aspect into account, it requires further study and field calibration. This is because in its present form it results in 43% tail loss during the wet season, @ 42 cumecs on average. Though it is primarily due to the consideration of social factors, i.e., necessity of bringing equity among users, which requires a minimum flow of 30 cumecs at the tail end to maintain sufficient head of water in the main canal to supply the downstream distribution channels, it requires further refinement. Further limitations of IOS include its inability to analyse the economic productivity of water, the most comprehensive of all the productivity measures (Small & Rimal 1996). This is because the economic value or the market prices of the crops are presently

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not taken into account while deciding which crop or field to irrigate. This may influence the crop production or economic return in the dry rabi season, when the water is scarce. However, this requires the development and calibration of a more comprehensive cost function to be used in the optimisation module.

It may also be pointed out that IOS is unable to explicitly handle the decisions regarding the area to be cultivated and crop types to be grown, especially in the dry rabi season. An additional simple optimisation scheme, a linear or goal-programming model, may be thought appropriate for the purpose, i.e., for replacing the present decision of arbitrarily cultivating 35% of the area with traditional crops. However, for solving such a scheme one would need to specify the distribution of water available during different crop growth stages as a constraint. Moreover, it will be difficult to account for the spatial variability of water along the canal length, which is possible only through a hydrodynamic model. Further, such a scheme would not be able to account for variations in soil moisture content, which is accomplished by MIKE SHE in IOS. Therefore, such an effort is not made at this stage. However, rabi simulation results have clearly shown the adaptability of IOS in water-short environment, thus, pointing to its versatility in handling wet or dry season. Therefore, in spite of its present limitations, IOS is potentially valuable for irrigation management in developing countries where the irrigation performance is far from satisfactory.

## Summary and conclusions

The broad details of the Irrigation Optimisation System (IOS), a decision support tool for helping the irrigation managers in planning and operation of large irrigation projects is outlined. IOS undertakes the hydraulic simulations of the canal system and hydrological simulations of the irrigated command simultaneously. The optimisation module maintains the canal releases close to the crop water demands and minimises the tail end losses. The irrigation scheduling module uses either the soil moisture or the water level approach and is well suited for a variety of crops, including paddy. The crop growth module calculates leaf area index, root growth, daily potential and maximum yields and yield loss due to moisture stress. The different modules in IOS interact with one another, providing useful information on various aspects of the irrigation canal command including canal losses, irrigation water utilisation, moisture status in the unsaturated zone and crop growth. IOS also provides the option of rotational schedule or on-demand irrigation with pre-specified or optimised releases. Though the rotational schedule is more popular in the developing countries, the on-demand irrigation option can help in developing the guidelines for the former.

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Results from the case study show that the application of IOS can lead to enhancements in crop production, irrigation intensity and physical productivity of water in large irrigation projects like MRP. IOS provides the irrigation managers, handling the traditional gravity irrigation projects in developing countries, with a versatile tool with which they can rationalise water use and help increase crop production.

Modelling of an entire irrigation command area using detailed simulation models of channel hydraulics, soil moisture and crop growth condition in combination with optimisation techniques is a complex matter requiring a solid data base and a powerful computer. Thus the main part of the computational resources in the IOS is spent by the MIKE 11 and MIKE SHE simulations. It may be questioned whether such advanced model codes indeed are required from a pure system optimisation point of view. If the purpose of the present study had been to focus on refinement of optimisation methods, e.g., include several control structures such as the cross regulators on the main canal in the optimisation computations, it would probably have been better, with today's computer technology, to use simpler simulation models than MIKE 11 and MIKE SHE.

However, contrary to most other tools presented in the literature, IOS purposefully focuses equally on the hydraulic and hydrological simulation models and the optimisation technique. In this way, it is possible to combine detailed information on the channel system and on soil moisture and crop growth at field levels with the optimisation technique. A key motivation for such an approach could be that the standard simulation models, typically utilised for planning and design studies in a system operation context, could be applied further when combined with standard optimisation techniques. The present paper has shown some of the possibilities, perspectives and problems of combining such methods, representing the state-of-the-art in their respective fields, into a comprehensive decision support system.

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# Water quality modelling at the Langerak deep-well recharge site

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ABSTRACT: This proceeding describes the application of a multi-species reactive transport model to a closely monitored deep-well artificial recharge test site at Langerak, The Netherlands. Using default parameter values for the processes and the measured native and recharge water quality parameters, it is possible to simulate the developments in the water quality very precisely.

## 1 INTRODUCTION

The water supply company South-Holland East (WZHO) supplies drinking water in the Rhine fluvial plain in the area between Amsterdam, Rotterdam, and Utrecht in the Netherlands. The main source of drinking water supply comes from bank-infiltrated Rhine River water that requires advanced and expensive purification techniques. At Langerak, The Netherlands, WZHO is investigating the use of a natural aquifer as an optional treatment plant for pretreated river water. The main objectives for WZHO is to investigate clogging effects and changes in the water quality during passage in the aquifer between an injection and a recovery well. The development of the content of trace metals is also in focus. The investigation is based on water quality observations from several multilevel wells and experiments conducted on the aquifer sediment. The data collection and experiments conducted were explicitly designed to enable a quantitative analysis of the change in native water quality due to the injection.

The quality of the water that is withdrawn from the aquifer depends on the quality of the native groundwater, the characteristics of the sediment and the quality and quantity of the injected water. Apart from the transport of solutes owing to the mixing of waters, several reactive processes also have to be accounted for in explaining the development of the water quality. The differences between the two solutions in oxidation level and inorganic cation concentration and the presence of various minerals in the aquifer will strongly affect the final quality of the water withdrawn in the recovery well. The following possible reactions are implicated: Flow and the related transport of solutes, redox processes, mineral dissolution/precipitation, complexation, and ion exchange.

A way to handle these processes quantitatively is in the framework of a reactive transport model, Which introduces the need for an accurate process formulation and the geometric distribution of the process parameters. Numerous chemical analyses are required, making it expensive to establish the distribution. The model application in this proceeding will try to evaluate the use of reactive transport models for describing and predicting the water quality changes at the Langerak site, predominantly using default parameter values known from geochemical model databases.

## 2 LANGERAK INJECTION SITE

Two wells, an injection and a recovery well, distanced by 190 m constitute the test field, see Figure 1.

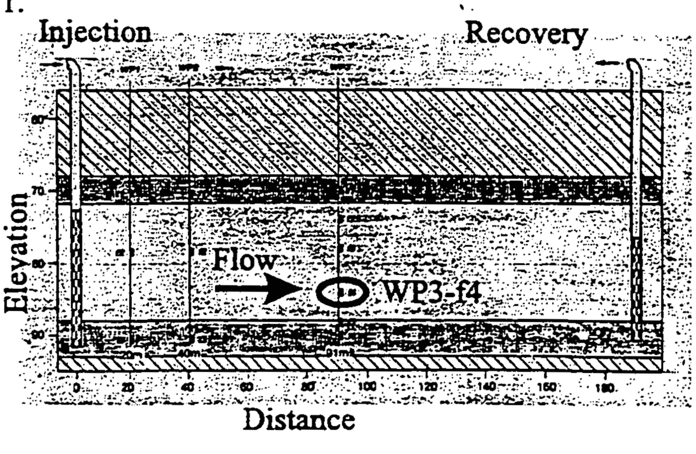


Figure 1: Schematised section of the Langerak test site. After Stuyfzand & Timmer 1998.

Between these wells, three monitoring wells are situated (WP1, WP2, and WP3) at distances of 20, 40, and 91 m from the injection well, respectively. Within each monitoring well, it is possible to conduct multilevel sampling from three to six differently elevated screens.

### 2.1 *Recharge water*

The injection experiment at the Langerak site took place during the period July 22, 1996 to July 22, 1997, one year in all. The water used for the recharge injection was local groundwater pre-treated by aeration and rapid sand filtering. Since the plan for future production is injection of pre-treated Rhine water, $NaNO_3$ was been added continuously, increasing the oxidation capacity of the injection water to allow it to become comparable to pre-treated Rhine water.

During the first month of the experiment, NaCl was added for tracer purposes in the attempt to determine the groundwater velocity and solute dispersion (Stuyfzand & Timmer, 1998). This initial NaCl pulse was injected during the period July 22 - August 22, 1996 and was succeeded by injection of plain water.

The mean of the measured concentrations of the major ions in the recharge water is shown in Table 1, along with the equilibrated concentrations used in the modelling. The recharge water is in an oxidized state, and the inorganic chemistry reflects these conditions. The recharge water is supersaturated with respect to calcite ($CaCO_3$).

Table 1: Total aqueous concentrations in the recharge water. (kgw: kg water).

| Species | Recharge water (Pulse) [mol/kgw] | Model input [mol/kgw] |
|---|---|---|
| pH | 7.7 | 7.7 |
| Pe | - | 12.85 |
| $Cl^-$ | $2.58\cdot10^{-3}$ ($3.05\cdot10^{-4}$) | $2.58\cdot10^{-3}$ ($3.05\cdot10^{-4}$) |
| $Ca^{2+}$ | $1.50\cdot10^{-3}$ | $1.50\cdot10^{-3}$ |
| $Mg^{2+}$ | $4.54\cdot10^{-4}$ | $4.54\cdot10^{-4}$ |
| $Na^+$ | $3.64\cdot10^{-3}$ ($1.58\cdot10^{-3}$) | $3.64\cdot10^{-3}$ ($1.58\cdot10^{-3}$) |
| $K^+$ | $1.67\cdot10^{-4}$ | $1.67\cdot10^{-4}$ |
| $Fe^{3+}$ | $7.23\cdot10^{-7}$ | $7.23\cdot10^{-7}$ |
| $Mn^{2+}$ | $1.65\cdot10^{-7}$ | $1.65\cdot10^{-7}$ |
| $NH_4^+$ | $8.36\cdot10^{-7}$ | $8.36\cdot10^{-7}$ |
| $O_2$ | $2.53\cdot10^{-4}$ | $2.527\cdot10^{-4}$ |
| $NO_3^-$ | $2.40\cdot10^{-4}$ | $2.40\cdot10^{-4}$ |
| $SO_4^{2-}$ | $7.96\cdot10^{-5}$ | $7.96\cdot10^{-5}$ |
| TIC | $5.11\cdot10^{-3}$ | $5.09\cdot10^{-3}$ |
| $HCO_3^-$ | $4.88\cdot10^{-3}$ | $4.88\cdot10^{-3}$ |

### 2.2 *Aquifer water and sediment*

The native aquifer is in an oxidative state that is more reduced than that of the recharge water. The reduction capacity is mainly associated with pyrite ($FeS_2$) and other reduced minerals (e.g., siderite ($FeCO_3$)), reduced exchanged cations ($NH_4^+$, $Fe^{2+}$, and $Mn^{2+}$), and methane ($CH_4$), see Table 2.

Table 2: Total aqueous concentrations in the native groundwater at WP3-f4. The X indicates exchanged species. Mineral concentration and exchanged species are given as their equivalent aqueous concentrations.

| Species | Native [mol/kgw] | Model input [mol/kgw] |
|---|---|---|
| pH | 7.3 | 7.16 |
| $Cl^-$ | $2.70\cdot10^{-4}$ | $2.70\cdot10^{-4}$ |
| $Ca^{2+}$ | $1.87\cdot10^{-3}$ | $1.84\cdot10^{-3}$ |
| $Mg^{2+}$ | $6.20\cdot10^{-4}$ | $6.03\cdot10^{-4}$ |
| $Na^+$ | $1.82\cdot10^{-4}$ | $1.82\cdot10^{-4}$ |
| $K^+$ | $1.90\cdot10^{-4}$ | $1.88\cdot10^{-4}$ |
| $Fe^{2+}$ | $8.71\cdot10^{-5}$ | $1.41\cdot10^{-5}$ |
| $Mn^{2+}$ | $1.20\cdot10^{-5}$ | $1.17\cdot10^{-5}$ |
| $NH_4^+$ | $5.20\cdot10^{-5}$ | $4.97\cdot10^{-5}$ |
| $CH_4$ | $3.90\cdot10^{-4}$ | $3.98\cdot10^{-4}$ |
| $NO_3^-$ | $< 7.00\cdot10^{-6}$ | $7.00\cdot10^{-6}$ as $N_2$ |
| $O_2$ | $< 3.00\cdot10^{-5}$ | 0.0 |
| $SO_4^{2-}$ | $\leq 1.20\cdot10^{-5}$ | $2.01\cdot10^{-12}$ |
| TIC | $8.17\cdot10^{-3}$ | $8.28\cdot10^{-3}$ |
| $HCO_3^-$ | $7.36\cdot10^{-3}$ | $6.94\cdot10^{-3}$ |
| NaX | - | $3.57\cdot10^{-4}$ |
| $CaX_2$ | - | $7.34\cdot10^{-3}$ |
| $MgX_2$ | - | $2.82\cdot10^{-3}$ |
| $FeX_2$ | - | $3.35\cdot10^{-5}$ |
| $MnX_2$ | - | $2.79\cdot10^{-5}$ |
| KX | - | $2.47\cdot10^{-4}$ |
| $NH_4X$ | - | $3.27\cdot10^{-5}$ |
| HX | - | $1.52\cdot10^{-7}$ |
| $CEC_{,X-}$ | | $2.00\cdot10^{-2}$ |
| Pyrite | $5.73\cdot10^{-3}$ | $5.73\cdot10^{-3}$ |
| Calcite | 0.227 | 0.227 |
| Goethite | 0.0 | 0.0 |

### 2.3 *Controlling reactive processes*

It is expected that the recharge of oxidized water will implicate oxidation of aquifer constituents, and the oxidized species will subsequently take part in other processes, e.g. precipitation of oxidized iron minerals here expressed by goethite (FeOOH). Oxidation of organic matter is here assumed insignificant owing to a low observed change in concentration of organic matter (not shown).

The different concentrations of the cations will influence the ion exchange complex. In particular, the high concentration of $Na^+$ injected as a pulse is expected to replace $Ca^{2+}$ and $Mg^{2+}$ initially bound in aquifer material. The subsequent normalisation will reverse the exchange. The ion exchanged concentr tions listed in Table 2 are based on equilibrium calculations using Gaines-Thomas coefficients from literature (Appelo & Postma 1993). pH is assumed to be controlled by the presence of calcite.

## 3 REACTIVE TRANSPORT MODEL

The flow and reactive solute transport are calculated using the MIKE SHE modelling system (Abbott et al. 1996). Three modules are used to compute the reactive solute transport:

1. The AD-module, which calculates three-dimensional solute transport.
2. The BM-module, which calculates kinetically controlled microbial-mediated degradation of organic matter under the influence of the surrounding geochemical environment.
3. The GM-module, which calculates reversible thermodynamically controlled equilibrium ion exchange, precipitation/dissolution, complexation, and redox processes. The GM module is based on the geochemical mass transfer model PHREEQC (Parkhurst 1995). It is modified slightly, so it can be coupled to the MIKE SHE system.

Since degradation of organic matter is assumed unimportant in these preliminary simulations, only the modules AD and GM are used when modelling the Langerak deep-well injection site, and hence only these will be discussed here. Several methods are available for solving the reactive transport equations (Engesgaard & Kipp 1992). In the MIKE SHE system, the solution to the coupled transport and the geochemical problem is carried out with a sequential non-iterative approach (SNIA). In SNIA, the AD-module solves the advection-dispersion equations in the first step hereby introducing a non-equilibrium state. In the following and final step, the GM module adjusts the transported concentrations to obtain equilibrium.

## 4 MODELLING THE LANGERAK SITE

At this preliminary stage, the modelling of the groundwater flow and the related transport will only take place in the framework of a one-dimensional (1D) model aquifer. The advantage of 1D simulations is a highly reduced calculation time. Among disadvantages are that variations in dilution, velocity, and dispersivity though the system, which are due to inhomogeneities in the aquifer and the flow pattern induced from the pumping system, are all neglected. The Langerak data support a fully three-dimensional (3D) model setup, but so far a 3D model has not been applied to the problem.

### 4.1 *Flow and transport setup*

The pore-velocity and the dispersivity are estimated by means of the analytical 1D model CXTFIT (Toride et al. 1995) for fitting the breakthrough curves for chloride for each screen. The velocity and dispersivity used in the following simulations are 0.9 m/day and 0.5 m, respectively. These parameters represent transport from the injection well to a specific screen WP3-f4, which is situated at well WP3, at level 86 m, see Figure 1. Comparison between measured and simulated concentrations will only be done for screen WP3-f4.

### 4.2 *Reactions setup*

Using an equilibrium assumption when modelling the reactive processes implies that the input solution and the native solution have to be charge-balanced and in equilibrium. Only the transport of solutes can disturb this equilibrium. The solution used in the model can therefore not be exactly matched since the measured concentration levels of the various species are not exactly charge-balanced or at equilibrium. The geochemical model does not accept, for example, that $O_2$ and $NH_4^+$ are present at the same time, since these constituents are not at chemical equilibrium. $O_2$ will be used to oxidize $NH_4^+$ until either $O_2$ or $NH_4^+$ is totally consumed. The equilibrium assumption explains the difference between the observed and modelled solution composition listed in Table 1 and 2. In addition, it has been necessary to treat $NH_4^+$ as an individual component not influenced by redox processes. If set at equilibrium, it will be instantaneously oxidized by $O_2$, $NO_3^-$ or $SO_4^{2-}$. It is included in the model as a cation exchanger only.

Default process parameters from the PHREEQC database are used with respect to complexation, redox and mineral processes. Partly fitted values are used only with respect to cation exchange reactions.

The mean of the measured solution concentrations in the recharge water is assumed to be at equilibrium with atmospheric $O_2(g)$ and $CO_2(g)$. The partial pressure of $CO_2$ has been slightly adjusted to make a better estimate of the modelled pH. In general, the calculated composition of the recharge water is comparable to the measured concentration. Only minor differences are observed in Table 1. The oxidizing capacity, associated with $O_2$ (4 eq./M), $NO_3^-$ (5 eq./M), and $SO_4^{2-}$ (8 eq./M), is approximately $2.85 \cdot 10^{-3}$ eq.

The background water as observed in WP3-f4 is set to be at equilibrium with the exchanger ($X^-$) and with minerals calcite and pyrite. The modelled reduction capacity, mainly associated with pyrite (15 eq./M), is $8.6 \cdot 10^{-2}$ eq. No direct measurement of CEC is available, so the exchanged concentrations are calculated on the basis of the concentration of the various ion exchangeable species. The selectivity coefficients used are based on literature values (Appelo & Postma 1993, Stuyfzand & Timmer 1998), see Table 3.

Table 3: Gaines-Thomas selectivity coefficients

| | |
|---|---|
| NaX | 1 |
| $CaX_2$ | 0.45 |
| $MgX_2$ | 0.5 |
| $FeX_2$ | 0.6 |
| $MnX_2$ | 0.55 |
| KX | 0.15 |
| HX | 1 |
| $NH_4X$ | 0.3 |

### 4.3 *Results and discussion*

Figure 2 shows the resulting breakthrough curves from the reactive transport simulation for several species. In all seven plots, solid or dashed lines represent simulated results whereas markers represent temporal measurements from the injection experiment. The location of the breakthrough is for all curves at WP3-f4. Looking first at the breakthrough curve for $Cl^-$, it is seen that the measured arrival and disappearance of the pulse is very well reflected in the simulation. Recall that the determination of the groundwater velocity and the dispersivity is based on this breakthrough curve. Thus, a good agreement between measured and simulated $Cl^-$ is almost obligatory. The background concentration is followed by the pulse, which is superseded by the recharge water concentration that is slightly higher than the background concentration. Only a few measurements are not exactly on the simulated line. $Cl^-$ broke through after approximately 100 days, and the pulse duration increased from the original 30 days to 40 days at 91 m downstream from the injection. It is worth noticing that approximately 100% $Cl^-$ mass recovery was found at this distance from the injection relative to the total input of $Cl^-$, indicating that the 1D assumption is acceptable. Both $HCO_3^-$ and pH are well predicted by the simulations. As the concentration of $HCO_3^-$ generally seems to slightly underestimate the measurements, measured and simulated pH are in good agreement, except for the background values of the aquifer at the beginning. The initial saturation of calcite could be said to "lock" initial pH to an erroneous result. With respect to the redox environment, the evolution of $CH_4$, $SO_4^{2+}$, and $Fe^{2+}$ is given. For $CH_4$, there is fair agreement between measured and simulated results, but the simulation seems to be insufficiently dispersed. The $SO_4^{2+}$ simulation tends to overestimate the measured concentration in the longer time scale corresponding to the recharge water. Similarly, it can be hard to recognise the concentration similarities between measured and simulated $Fe^{2+}$. The poor fit of $Fe^{2+}$ and overestimation of $SO_4^{2+}$ are believed to be caused by an only partial oxidation of pyrite and perhaps a presence of siderite. After one year of travel, oxidized conditions have still not bee obtained in the aquifer at this distance. $O_2$ and $NO_3^-$ are not shown, since no changes in concentration occur. Thus, $O_2$ and $NO_3^-$ are strongly retarded relative to $Cl^-$.

The chromatographic behaviour of the cation exchange is illustrated by the breakthrough curves of the cations. In general, there is good accordance between the simulated and measured cation concentrations. As the $Cl^-$ pulse passes, the $Na^+$ concentration increases as well. $Ca^{2+}$, $Mg^{2+}$, and $K^+$ decrease to a level lower than both background and recharge water. As $Na^+$ replaces the other cations on the exchange complex, a release is observed. However, as the concentration of $Na^+$ decreases again, mass from the aqueous phase of the other cations will be taken to cover the loss of the exchange complex. $Mn^{2+}$ is also well described b the simulation, although slightly overestimated. manganese substitution in siderite was suggested by Stuyfzand & Timmer (1998) as an explanation for the deviations. Unfortunately, the simulation of $NH_4^+$ does not yield a picture similar to these measurements. All cations break through a little later than $Cl^-$, indicating a minor retardation. It is seen that $Na^+$ is slightly retarded, $Ca^{2+}$ and $Mg^{2+}$ are more retarded, and $K^+$ is even more retarded. According to the measured data, $NH_4^+$ should be strongly retarded. Because of ion exchange competition, the shape of the retarded species is not perfectly Gaussian. The retarded species recover more slowly to the recharge concentrations characterised by more dispersed breakthrough curves relative to chloride.

The evolution of minerals pyrite and calcite is shown as the difference between the actual concentration and the native mineral concentratio Pyrite starts precipitating at the time of the breakthrough of $Cl^-$. Near the injection well, pyrite is oxidized, and $Fe^{2+}$ and $SO_4^{2-}$ are dissolved as the aquifer becomes oxidized by the recharge water. Downstream, the increased concentrations of $Fe^{2+}$ and $SO_4^{2-}$ make pyrite precipitate again, because the aquifer at downstream locations still is reduced. After the breakthrough of $Cl^-$, the concentration of pyrite increases to a constant level at equilibrium with the incoming $Fe^{2+}$ and $SO_4^{2-}$ because of the constant upstream pyrite oxidation rate. The concentration will remain constant until the retarded oxidation front arrives at the well, but this had not happened within the 365 days of the experiment. Calcite starts to be dissolved when $Cl^-$ breaks through owing to the lower content of $HCO_3^-$ in the recharge water. Because $Ca^{2+}$ contributes to th cation exchange process, the $Na^+$ pulse is reflected i calcite dissolution as well, which explains the lack of dissolved $Ca^{2+}$ in the cation breakthrough curve.

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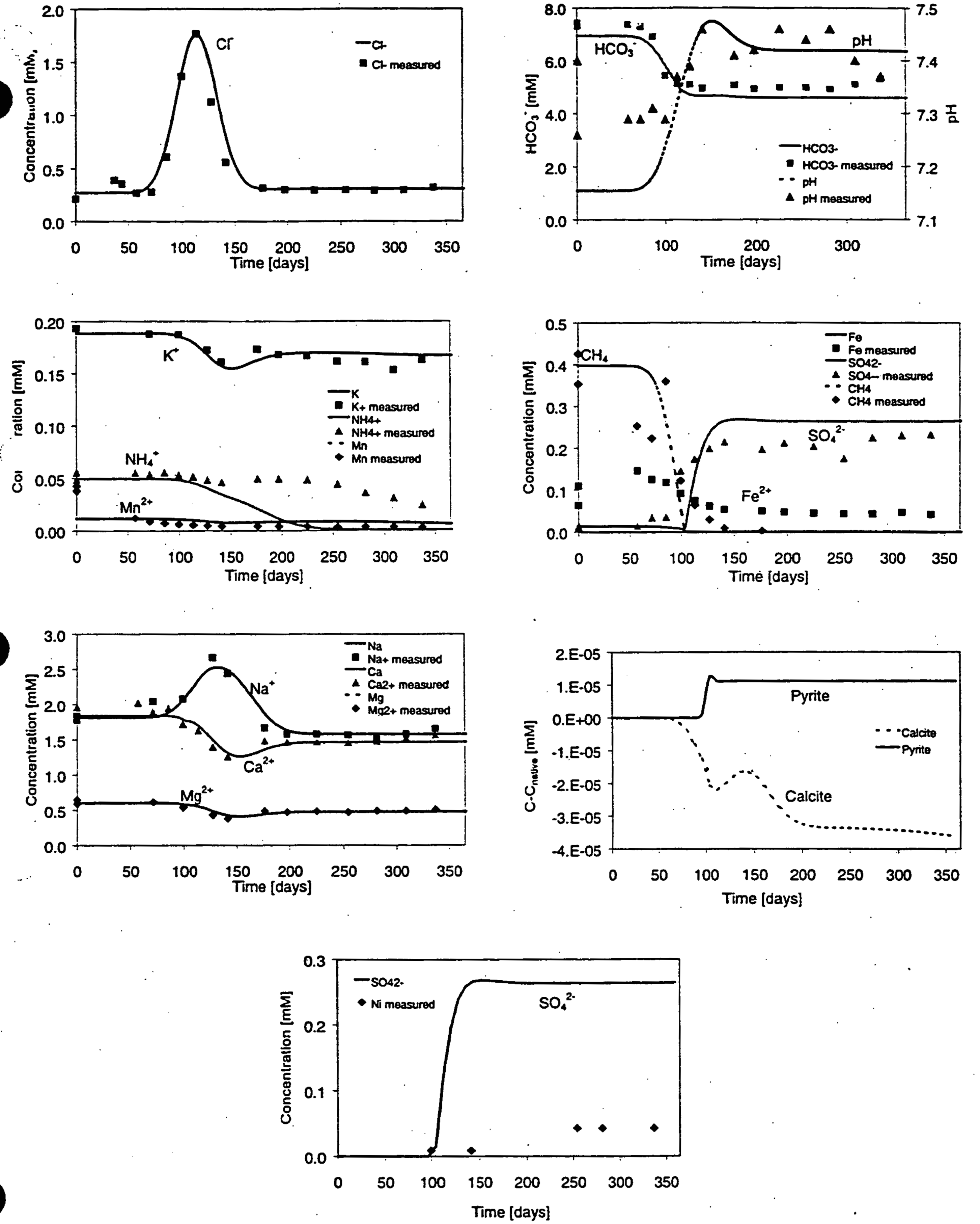


Figure 2: Simulated (lines) and observed (markers) breakthrough curves of numerous species for screen WP3-f4.

The measured concentrations of nickel (Ni) and $SO_4^{2-}$ are also illustrated. Ni is probably substituted in the pyrite and consequently dissolved as pyrite is oxidized. The same picture can be observed for other trace minerals like arsenic, cadmium, and zinc (not shown). It is expected that when the pyrite has been totally oxidized, a precipitation of $Fe^{3+}$-minerals may constitute a base for trace mineral sorption. However, this state of aquifer development had not been reached at WP3 within the test period.

In general, a good agreement between the measured and simulated concentrations is seen despite the simplification by using only 1D and default values from a chemical data base.

A future application of 2D and 3D flow patterns will probably improve the results with respect to fitting the breakthrough curves from all screens. None of the investigated species constitute a danger to human health in the concentrations developed. However, an inclusion of the transport of trace metals associated with pyrite and goethite will justify the modelling in this sense. Clogging effects owing to mineral precipitation should also be analysed. Sorption of organic micropollutants (e.g., pesticides) to precipitated iron minerals will be investigated. In the end, the oxidation of pyrite will finally be complete, and the aquifer will be entirely oxidized. Thus, there is an upper limit of the reduction capacity of the aquifer. Likewise, if the artificial recharge at the Langerak site is halted, a re-reduction of the aquifer will take place. Sorbed trace metals and organic micropollutants will be desorbed, with consequences for a future withdrawal. Reactive transport simulations can, in this respect, be a powerful tool for investigating the future behaviour of the aquifer development independent of human choices.

## 5 CONCLUSION

Owing to time constraints, not all of the information available at the Langerak deep-well injection site has been incorporated into this model exercise. The results presented here are focused on the observations from a single screen in one of the three wells. The observations concerning the other screens are from a reactive transport viewpoint somewhat analogous to the observations in the screen used in this study. The difference is believed to be caused by different travel times from the injection point to the various screens and the lack of equilibrium in the screens closest to the injection point. Hopefully the promising results from the one-dimensional model aquifer can be verified when the site is modelled in several dimensions. The setup of the model has been based on observations of the quality of the recharge water and the native aquifer, the observation of the breakthrough of a conservative tracer, and default parameters for the reactive parameters. From these observations, it has been possible to simulate the major geochemical processes occurring in the aquifer and the related change in water quality precisely and quantitatively. On the basis of the work carried out until now at the Langerak site, it is concluded that a reactive model is a valuable tool in the design and operation of an artificial recharge plant, at least for this specific site. Whether this is also the case for other aquifers with other physical and geochemical properties has yet to be proven.

## 6 ACKNOWLEDGEMENT

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# Integrated Hydrological Modelling in South Florida Water Management District

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**Abstract**
A physically based, integrated hydrological modelling system is used by the South Florida Water Management District (SFWMD) to address water management issues involving interactions between ground water and surface water flow regimes. The modelling system is based on the MIKE SHE *(Refsgaard and Storm, 1995)* and MIKE 11 *(Havnø and Madsen, 1995)* systems developed by the Danish Hydraulic Institute (DHI). The MIKE SHE/MIKE 11 modelling system is being further developed and tailored to South Florida conditions, as part of the project "Small Scale Surface Water – Ground Water Model" being conducted by SFWMD. This paper provides a brief description of the applied MIKE SHE and MIKE 11 modelling systems and of the enhancements being carried out as part of the SFWMD project. Preliminary results of the first MIKE SHE application at the SFWMD are presented.

## Introduction
The South Florida Water Management district is home to one of the largest wetland systems in North America, stretching southward from the Kissimmee River to Lake Okeechobee to the Everglades and finally to Florida Bay. Historically, this wetland spilled out of Lake Okeechobee as a river about 50 miles wide and 200 miles long, but only a few inches deep. In the first half of this century, canals were dug, and levees built to contain the marsh and drain the land for residential and agricultural development. Recently, however, there has been a move to restore parts of the Everglades system to more historic conditions. Work is proceeding to allow over bank spilling, channel meandering and reduction of impacts from over exploitation of ground water. The intricacy of the wetland flow system and complex interactions between surface water, ground water and

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evapotranspiration, in combination with massive human interventions combine to make the Everglades restoration project a formidable technical problem. Its solution will require application of advanced mathematical modelling tools.

**Role of Integrated Hydrological Modelling in SFWMD**

Exploitation of water always affects the hydrological cycle. Overexploitation of ground water for public water supply, irrigation and industry can lead to destruction of wetlands, lakes and rivers. An integrated hydrological model can predict effects of human interventions on the hydrological cycle (e.g. the minimum discharge in a river, or the water level in a lake, the extent of a wetland etc.) and thereby create a more informed basis for policy and decision making. For SFWMD the most obvious application areas lies within general water resources management and policy making focusing, in particular, on wetland protection and restoration, public water supply and irrigation.

**The MIKE SHE Modelling System**

MIKE SHE is a deterministic, fully distributed, and physically based modelling system for describing the major flow processes of the entire land phase of the hydrological cycle (see Fig. 1). The MIKE SHE model was derived from the SHE model *(Abbott et al., 1986a and 1986b).* SHE was developed during the 1970's as a joint European effort of the University of Newcastle, Laboratoire National d'hydraulic de France (French National Hydraulic Laboratory) and DHI. The concept and philosophy behind MIKE SHE is basically the same as for SHE. The main differences are in regard to user friendliness, pre- and post-processing facilities, code efficiency and robustness. Moreover a number of add-on modules have been developed for MIKE SHE for instance for pollution transport, geochemistry and biology. The basic water flow module of MIKE SHE includes the following processes:

*Interception and evapotranspiration:* The interception storage is expressed as a function of Leaf Area Index (LAI). The actual evapotranspiration may be calculated using the empirical Kristensen and Jensen model *(Kristensen and Jensen 1975)*, or the Penmann-Monteith model.

*Overland Flow:* The overland flow module is a 2-dimensional final difference model using a diffusion wave approximation of the Saint Venant equations. The overland flow module is fully coupled to the river module and may act both as a source and a sink for river flow.

*Channel Flow:* The original version of MIKE SHE used a 1-dimensional diffusion wave approximation of the Saint Venant equations. This river module has a number of shortcomings that limit its applicability for studies involving complex surface water regimes. For instance it does not support hydraulic structures and rivers were considered as a line located between model grids. Consequently rivers could not be wider than one model grid.



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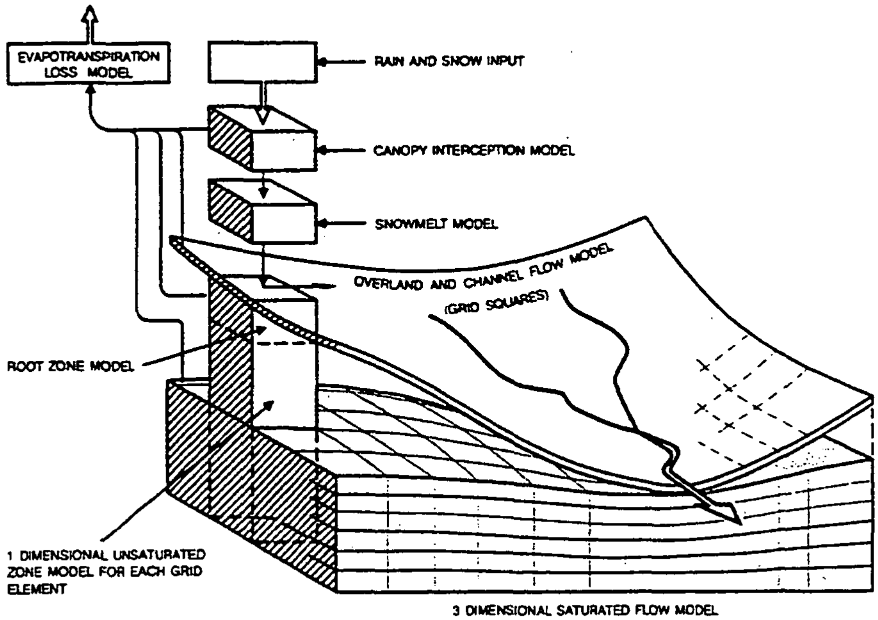


Figure 1. Structure of the MIKE SHE Modelling system

As part of the SFWMD project the original river module of MIKE SHE is substituted with the far more advanced MIKE 11 modelling system.

*Unsaturated flow:* In the most comprehensive mode MIKE SHE solves the full Richard's equation. A simplified version that neglects the tension term of Richard's equation is also available.

*Saturated flow:* The saturated zone module is a traditional 3-D final difference ground water model. In structure and flexibility the ground water flow model is similar to the USGS MODFLOW model.

**The SFWMD Project**

The SFWMD Project was launched in July 1997. The main objectives are to develop and test an integrated hydrological modelling system for SFWMD. The original MIKE SHE, described above, is being tailored to South Florida conditions. The model development phase ends March 1998 and is to be followed by a testing phase that ends in September 1998. During the testing phase, the modified modelling system will be applied to simulate the hydrological functions of the Everglades Nutrient Removal site located southeast of Lake Okeechobee. As part of the project MIKE SHE is being interfaced to SFWMD databases and GIS applications to facilitate model construction and simplify conversion between MIKE SHE and existing SFWMD models. In addition, a number of new modules and improvements are implemented. These are briefly described below:



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*Integration of the MIKE 11 river modelling system with MIKE SHE*

In an area like South Florida, with small hydraulic gradients and numerous hydraulic control structures, a fully dynamic river model is required in order to simulate water level variations with any precision. A river model must also be able to account for a range of different hydraulic structures to meet SFWMD needs. The hydrodynamic (HD) module of MIKE 11 meets these requirements. A coupling between MIKE SHE and MIKE 11 HD was originally developed by DHI as part of a major project in the Slovak republic *(DHI, 1995)* and subsequently tested in connection with other model applications. Since the concept and methodology behind the MIKE SHE/MIKE 11 integration are well established, the developments which take place under the SFWMD project focus primarily on making a generic, user friendly and numerically robust coupling of the two modelling systems.

*Simple Evapotranspiration and Infiltration Module*

As an alternative to MIKE SHE's more complex modules a new simple module is being implemented based on a formulation by *Yan and Smith (1994).* This module considers the root zone as just one layer and calculates actual evapotranspiration from interception storage, ponded water, moisture content in the root zone and evaporation from the saturated zone using a MODFLOW extinction depth approach.

*Irrigation and Crop Growth Module*

The irrigation module calculates irrigation demand based on water availability in the root zone. The module uses the so-called MAD concept (Maximum Allowable Deficit), where irrigation water is applied whenever the water content drops below crop dependent threshold values. Irrigation water may be supplied from ground water as well as surface water, and may be applied as sprinkler irrigation or sheet irrigation. Crop growth may be prescribed in terms of leaf area index and root depth, or it may be calculated using a crop-growth module. The crop growth module uses FAO crop yield coefficients *(FAO, 1979)* and uses an s-shaped, Gompertz growth equation *(Thornley, 1976. Thornley and Johnson, 1990).*

**Alligator Lake Draw Down Study**

The Alligator Lake draw down study is the first modelling study in which the new integrated model is applied. In order to harvest undesired water plants in the shallow parts of Alligator Lake, the SFWMD has plans to draw down the water level in the lake from the normal 19.4 m to 17.6 m. A number of tropical fish farms are located in the vicinity of the lake. The fish farmers are concerned that the lake level may affect the ground water table in the surficial aquifer system. If the ground water level drops, the water level in the fishponds may drop accordingly. Significant drops in fish pond levels could result in large economic losses for the fish farmers. The objectives of the study are as follows: 1) to test if the model is suitable for simulating the Florida hydrological conditions, 2) evaluate if the lake draw down will have impacts to the groundwater levels near the fish ponds. If the model passes the test, the model will be used to assist water managers to schedule and design the draw down operations for minimising the impacts, if any, to the groundwater tables in the vicinity of the fishponds.

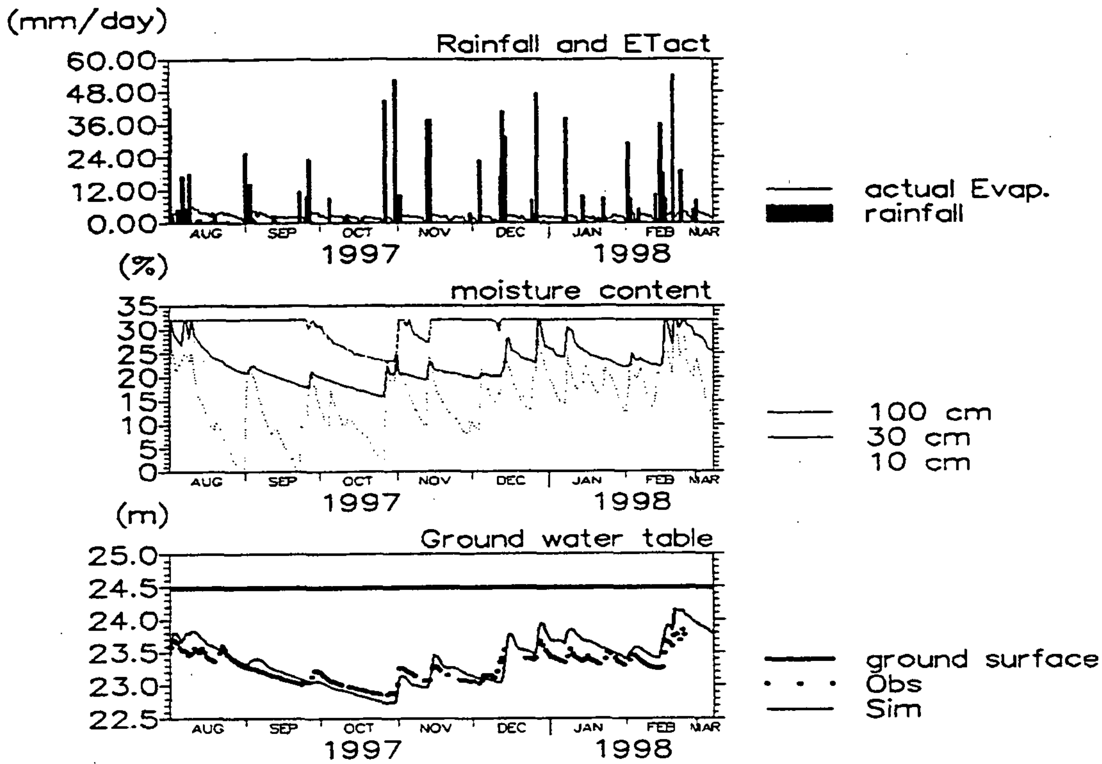


*Fig. 2 Time-series of rainfall, simulated actual evapotranspiration, moisture content in unsaturated zone and ground water table. (The moisture content plot shows results in 3 different depths. The soil is a Myakka soil with saturated water content=32%, field capacity=14%, and wilting point=3%)*

According to the draw down schedule, the water level will be kept on 17.6 m for about 3 months. Subsequently, all gates that divert water from the lake will be closed, and the water level will start to recover. The recovery of the lake water level will depend on weather conditions. There are concerns that this recovery may take several months or even longer. If the water level is maintained at 17.6 m for a long period of time, it may ultimately affect the ground water table significantly. In order to determine the recovery period under different conditions, the model will be used to simulate some water level recovery scenarios, for instance assuming a wet year, a typical year and a dry year. The study was launched in November 1997 and is being carried out as a joint effort between DHI and the SFWMD. DHI established a roughly calibrated model for SFWMD and the final model calibration, validation and scenario simulations are done by SFWMD staff, with assistance from DHI. Preliminary model results indicates that the model performs reasonably well (Fig. 2). The figure shows rainfall, evapotranspiration, soil moisture content in three different depths and simulated and observed ground water level, at a location near one of the fish farms.

**Conclusions**

An integrated hydrological modelling system has been developed based on DHI's MIKE SHE and MIKE 11 systems. The integrated modelling system provides capabilities to simulate the entire hydrological cycle. The first model in the Lake Alligator area in South-Florida, indicates that the model functions correctly, and has passed the preliminary test for simulating the South Florida hydrological conditions.

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*Invited overview presentation at 2nd BALTEX Study Conference*

# CONCEPTUAL VERSUS PHYSICALLY-BASED HYDROLOGICAL MODELS: WHICH MODELS TO BE USED FOR BALTEX PURPOSES ?

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## Introduction – hydrological modelling approaches

The two traditional approaches in hydrological modelling are the conceptual and the physically-based ones, as illustrated by e.g. the HBV (Bergström, 1995) and the MIKE SHE (Refsgaard and Storm, 1995). The conceptual models are used in either a lumped or a semi-distributed mode. A lumped model implies that the catchment is considered as one computational unit, whereas a semi-distributed model uses some kind of distribution, either in subcatchments or in hydrological response units, where areas with the same key characteristics are aggregated to sub-units without considering their actual locations within the catchment. Examples of hydrological response units considered in semi-distributed models are elevation zones, which are relevant for snow modelling, and combinations of soil and vegetation type, which may be relevant for evapotranspiration (and hence runoff generation) modelling. A distributed model, on the other hand, provides a description of catchment processes at georeferenced computational grid points within the catchment. A physically-based model may be used as either fully distributed or, in certain cases, semi-distributed.

The fundamental difference between the conceptual and the physically-based models lies in their process descriptions and the way spatial variability is treated. The physically-based models contain equations which have originally been developed for point scales and which provide detailed descriptions of flows of water, solutes and energy. The variability of catchment characteristics is accounted for explicitly through the variations of hydrological parameter values among the different computational grid points. This approach leaves the variability within a grid as un-accounted for, which in some cases is of minor importance but in other cases may pose a serious constraint. The conceptual models (irrespective of whether they are used in a lumped or a semi-distributed mode) uses empirical process descriptions, which have built-in accounting for the spatial variability of catchment characteristics. The conceptual models have been developed for and are very good at describing runoff generation and overall water balances, but generally fail to provide details of water flows and soil moisture storages as well as descriptions of fluxes of solutes and energy.

Several intercomparison studies of lumped conceptual, semi-distributed conceptual and distributed physically-based models have indicated that their respective capability of predicting catchment runoff are almost identical (e.g. Refsgaard and Knudsen, 1996). The question of which model type to use for BALTEX purposes thus boil down to the question of what the purpose of the hydrological modelling is. Hence for keeping track of runoff and water balances the conceptual models are adequate and suitable. However, if the purpose is extended with a wish to provide descriptions of the spatial pattern of soil moisture and energy fluxes, then the only option left is the physically-based models.

The HBV conceptual model has already successfully been established for runoff simulations of the Baltic Sea catchment (Bergström and Carlsson, 1994). The key data used for this exercise are hydrometeorological time series (precipitation, temperature, potential evapotranspiration for a large number of stations). To use physically-based models at the same scale requires the following two problems to be solved:

- Readily available data from national, regional or global databases must be used. In addition to hydrometeorological data also used by conceptual models, data on topography, soil type, land use and geology must be available. In order to use a physically-based model at such large scale in practise these additional data must be readily available in digital form.
- A consistent methodology for upscaling/aggregation of physically-based models for use at the BALTEX scale is required. Due to the large catchment area the computational grid sizes will inevitably be so large (1 $km^2$ or larger) that it is impossible explicitly to resolve all the variability of catchment characteristics.

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Therefore, the upscaling/aggregation methodology must somehow also consider the variability existing within each grid area.

The presentation will discuss the possibilities of overcoming these two problems and present examples of MIKE SHE applications to large scale modelling.

## Global and regional databases

Digital global and regional databases of interest for large scale hydrological modelling such as in BALTEX are rapidly becoming available in these years. Examples include 1x1 $km^2$ DEM's available over the Internet and the GISCO database containing amongst others soil type and land cover for most of Europe. Together with suitable transfer functions they can provide very useful input data for large scale modelling. The key gap generally remaining when looking for readily available global data appears to be geological data. Whereas inclusion of geological data generally is very important, these data are not so crucial for the rather surface oriented research carried out within BALTEX.

## Upscaling of physically-based models for application to BALTEX scale

The complex interactions between spatial scale and spatial variability is widely perceived as a substantial obstacle to progress in this respect (Blöschl and Sivapalan, 1995; and many others). Often a distinction is made between the terms upscaling and aggregation. Thus, upscaling is a special case of spatial aggregation, namely one in which the objective is to transform the point parameter values into 'effective' block parameter values, such that the microscale equations in the model become valid at the macroscale. A principal difference between aggregation and upscaling is that whereas aggregation can be defined irrespective of a model operating on the aggregated values, upscaling must always be defined in the context of a model that uses the parameters that have been scaled up (Heuvelink and Pebesma, submitted)

The research results on the scaling issue reported during the past decade have, depending on the particular applications, focussed on different aspects, which may be categorised as follows:

- subsurface processes focussing on the effect of geological heterogeneity;
- root zone processes, including interactions between land surface and atmosphere; and
- surface water processes focussing on topographic effects and stream-aquifer interactions.

The effect of spatial heterogeneity on the description of subsurface processes has been the subject of comprehensive research for two decades, see e.g. Dagan (1986) and Gelhar (1986) for some of the first consolidated results and Wen and Gómez Hernández (1996) for a more recent review. The focus in this area is largely concerned with upscaling of hydraulic conductivity and its implications on solute transport and dispersion processes in the unsaturated zone and aquifer system, typically at length scales less than 1 km.

The research in the land surface processes has mainly been driven by climate change research where the meteorologists typically focus on length scales up to 100 km. Michaud and Shuttelworth (1997), in a recent overview, conclude that substantial progress has been made for the description of surface energy fluxes by using simple aggregation rules. Sellers et al. (1997) conclude that "it appears that simple averages of topographic slope and vegetation parameters can be used to calculate surface energy and heat fluxes over a wide range of spatial scales, from a few meters up to many kilometers at least for grassland and sites with moderate topography". An interesting finding is the apparent existence of a threshold scale, or representative elementary area (REA) for evapotranspiration and runoff generation processes (Wood et al., 1988; Wood et al. 1990; Woods et al., 1995). Famiglietti and Wood (1995) concludes on the implications of such an REA in a study of catchment evapotranspiration that "the existence of an REA for evapotranspiration modelling suggests that in catchment areas smaller than this threshold scale, actual patterns of model parameters and inputs may be important factors governing catchment-scale evapotranspiration rates in hydrological models. In models applied at scales greater than the REA scale, spatial patterns of dominant process controls can be represented by their statistical distribution functions". The REA scales reported in the literature are in the order of 1 – 5 $km^2$.

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The research on scale effects related to topography and stream- aquifer interactions has been rather limited as compared to the above two areas. Saulnier et al. (1997) has examined the effect of the grid sizes in digital terrain maps (DTM) on the model simulations using the topography-based TOPMODEL. They concluded that in particular for channel pixels the spatial resolution of the underlying DTM is important. Refsgaard (1997) using the distributed MIKE SHE model to the 440 $km^2$ Danish Karup catchment with grid sizes of 0.5 km, 1 km, 2 km and 4 km, found that the discharge hydrograph shape was significantly affected for the 2 and 4 km grids as compared to the almost identical model results with 0.5 and 1 km grids. He concluded that the main reason for this change was that the density of smaller tributaries within the catchment was smaller for the models with the larger grids.

Many researchers doubt whether it is feasible to use the same model process descriptions at different scales. For instance Beven (1995) states that "... the aggregation approach towards macroscale hydrological modelling, in which it is assumed that a model applicable at small scales can be applied at larger scales using 'effective' parameter values, is an inadequate approach to the scale problem. It is also unlikely in the future that any general scaling theory can be developed due to the dependence of hydrological systems on historical and geological perturbations." Similarly, Wen and Gómes Hernández (1996) emphasise the fundamental problems in the different upscaling techniques.

The research within BALTEX mainly deals with scaling of root zone processes, while the subsurface processes and stream-aquifer interaction play minor, although not uninteresting, roles. A key challenge as compared to the experiences reported in the literature is then how to make use of the physically-based model at large scale without possibility for detailed calibration at that scale, when we know that its physically-based equations are developed for small scales. Such model can only be stated as well proven for small scales, and the few attempts made so far to use it on scales above 1,000 $km^2$ have applied calibration at that scale (e.g. Refsgaard et al. 1992; Jain et al., 1992). The basic upscaling methodology proposed for this purpose is based on a semi-distributed approach:

- The basic modelling system is of the distributed and physically-based type. For application at *point scale* (where it is not used spatially distributed) the process descriptions of this model type can be tested directly against field data.
- The model is used with (equations and) parameter values in each horizontal grid point representing *microscale* conditions. The microscale is selected as a scale for which previous tests have indicated that the model is able to describe the basic processes satisfactorily (50 – 200 m).
- The smallest horizontal discretization in the model is the *grid scale* or grid size (1 – 5 km) that is larger than the microscale. This implies that all the variations between categories of soil type and crop type within the area of each grid can not be resolved and described at the grid level. Input data that vary at microscale, and whose variations are not included in the grid scale model representation, are distributed at the macroscale so that their statistical distributions are preserved at that scale.
- The results from the microscale modelling are then aggregated to *macroscale* (10 – 50 km) and the statistical properties of model output and field data are then compared at macroscale.
- For applications to larger scales than macroscale, such as *BALTEX scale*, the macroscale concept is used, just with more grid points. This implies that the BALTEX scale can be considered to consist of several macroscale units, within each of which the microscale statistical variations are preserved and at which scale the predictive capability of the model thus lies.

The microscale considered is not point scale, but rather a field scale characterised by 'effective' soil and vegetation parameters, but assuming only one soil type and one vegetation type. Thus the spatial variability within a typical field is aggregated and accounted for in the 'effective' parameter values. Physically-based models have previously at many occasions documented their ability to describe conditions at field scale by use of effective parameters (e.g. Jensen and Refsgaard, 1991a,b,c). In the aggregation to macroscale the variations among soil types and crops are preserved statistically, although they are not correctly georeferenced. A field scale is typically 1 ha, whereas the model grids used are 2-3 orders of magnitude larger. The macroscale may be an entire catchment, but would for larger scale representations such as the Baltic Sea catchment be subareas of maybe 100 model grids. This implies that the macroscale model does not pretend to provide a correct description at a grid scale, because the within-grid variation of soil and vegetation types are not described at that scale. However, at the macroscale, the statistical properties of the model aim at describing well the statistical properties of field conditions.

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### Conclusion

Both the conceptual and the physically-based models are required for BALTEX. The two model types have different capabilities, potentials and limitations:

- The conceptual models are today ready for operational use for simulation of runoff at the BALTEX scale. Thus for establishing overall water balances for the Baltic Sea catchment this model type can provide significant contributions. The conceptual models can easily be semi-coupled with atmospheric models in such a way that the atmospheric model generates input for the hydrological model, but due to incompatible spatial resolution and lack of energy balance description it is not possible to make a suitable feedback mechanism to the atmospheric model.
- The physically-based models enable a more detailed description of hydrological processes with a better spatial resolution, which is of great importance for the research on land surface-atmosphere interactions. Thus this model type makes it possible to simulate spatial land surface patterns which are directly compatible with available spatial remote sensing data. Because physically-based models contain spatial information on soil moisture and easily enables inclusion of standard SVAT schemes, they can be fully coupled with atmospheric models with mutual feedback mechanisms.

In brief, it may be concluded that at present the conceptual models comprise a major operational potential, whereas the distributed models have the major research potential of interest in a BALTEX context.

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# THE EUROPEAN SOIL EROSION MODEL (EUROSEM): A DYNAMIC APPROACH FOR PREDICTING SEDIMENT TRANSPORT FROM FIELDS AND SMALL CATCHMENTS

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## ABSTRACT

The European Soil Erosion Model (EUROSEM) is a dynamic distributed model, able to simulate sediment transport, erosion and deposition over the land surface by rill and interill processes in single storms for both individual fields and small catchments. Model output includes total runoff, total soil loss, the storm hydrograph and storm sediment graph. Compared with other erosion models, EUROSEM has explicit simulation of interill and rill flow; plant cover effects on interception and rainfall energy; rock fragment (stoniness) effects on infiltration, flow velocity and splash erosion; and changes in the shape and size of rill channels as a result of erosion and deposition. The transport capacity of runoff is modelled using relationships based on over 500 experimental observations of shallow surface flows. EUROSEM can be applied to smooth slope planes without rills, rilled surfaces and surfaces with furrows. Examples are given of model output and of the unique capabilities of dynamic erosion modelling in general. © 1998 John Wiley & Sons, Ltd.

KEY WORDS: soil erosion model; soil erosion processes; distributed modelling; dynamic modelling.

## INTRODUCTION

An increasing awareness by scientists, governments and the general public that soil erosion is an important problem within the countries of the European Community (Morgan and Rickson, 1990) has drawn attention to the lack of a satisfactory system in Europe for assessing the risk of erosion, predicting erosion rates and designing and evaluating different soil protection strategies. Present technologies for erosion assessment, based on scoring systems for rainfall erosivity, soil erodibility, slope and land use (Auerswald and Schmidt, 1986; Rubio, 1988; Briggs and Giordano, 1992; Jäger, 1994), provide information on the spatial distribution of erosion risk but only limited data on erosion rates. Attempts to use the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) in Europe as a technique for predicting erosion rates and evaluating different soil conservation practices show that great care is required in the selection of input values for rainfall ($R$) (Chisci and Zanchi, 1981; Richter, 1983) and soil erodibility ($K$) (Richter, 1980; De Ploey, 1986; Schwertmann, 1986) factors. Also, the equation is of limited value since it cannot provide information on the fate of sediment once it is eroded. The USLE is not able to predict deposition or the pathways taken by eroded material as it moves from hillslope sites to water bodies. In a European context, where the most important consequences of erosion are pollution and sedimentation downstream rather than loss of productivity on-site, policy-makers need to know more about the location of sediment sources and sinks. Further, the design of strategies to control pollution associated with runoff and erosion on agricultural land requires knowledge of what happens in individual rain

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storms, often on a minute-by-minute basis, in order to predict the size and timing of peak discharges of water and sediment from hillslopes to rivers. The USLE cannot provide this because it predicts only mean annual soil loss.

In order to provide a better representation of erosion processes, American scientists have concentrated in recent years on developing more physically based erosion models such as those used in CREAMS (Knisel, 1980; Foster *et al.*, 1981), ANSWERS (Beasley *et al.*, 1980), KINEROS (Woolhiser *et al.*, 1990) and WEPP (Nearing *et al.*, 1989). Similar models are also being developed in Australia (Rose *et al.*, 1983; Misra and Rose, 1992).

Whilst both CREAMS and WEPP can be run for individual storms, they simulate only total storm soil loss, and assume a steady surface flow profile. They do not model peak sediment discharge or describe the pattern of events within a storm, or provide a sediment graph showing the pattern of sediment discharge over time, information which is useful for looking at potential pollution loadings from sediment fluxes into water courses. For catchments where one or two storms account for most of the annual soil loss, which is the typical situation in many European countries (Sfalanga and Franchi, 1978; Boschi ahnd Chisci, 1978; Richter, 1979; Raglione *et al.*, 1980; Boschi *et al.*, 1984; Tropeano, 1984; Chisci *et al.*, 1985; Govers, 1987), steady flow is rarely achieved, and the CREAMS and WEPP methodologies may be inappropriate. In order to obtain a good approximation of what is happening over the land surface in such events, a fully dynamic approach to erosion modelling is required.

The need for an alternative approach was recognized at the European Community Workshop held in Brussels, 1986, when Chisci and Morgan (1988) proposed a framework for a European model to be based on the best European research into erosion processes and their control. One of the recommendations of the Workshop was that European scientists should 'try to develop a new general erosion model for use in the EC countries for erosion risk evaluation and the design of erosion control measures' (Chisci, 1988). This paper describes the outcome of the resulting research effort which has led to the European Soil Erosion Model (EUROSEM). In addition to describing EUROSEM, emphasis is given to the features which make it different from and an advance upon the American and Australian models referred to above.

## APPROACH

Since soil erosion by water is closely related to rainfall and runoff, erosion modelling cannot be separated from the procedures used to model the generation of runoff and its routing down a hillside and through the river channel network. Some models, like WEPP and CREAMS, use continuous simulation to model the generation of runoff. Continuous simulation models require a large amount of input data for weather and land use that are only indirectly related to erosion studies. In addition, they are sensitive to modelling evapotranspiration and soil dynamics, and simulate a large number of small events that may not produce significant runoff or soil loss. It seems therefore more appropriate to focus on the development of an event-based model, considering, as indicated above, that erosion is dominated by only a few events per year which are characterized by a highly dynamic behaviour.

An alternative approach was therefore adopted based on simulating the dynamic behaviour of events within a storm. Within-storm modelling is also more compatible with the equations used in process-based models to describe the mechanics of erosion. These equations are strictly applicable to instantaneous conditions and they cannot be applied to average conditions without loss of accuracy. Applying them to conditions averaged over one minute is thus more acceptable than using them for conditions averaged over one hour or more. It is recognized, however, that this approach creates problems for the determination of certain input data, particularly relating to the starting conditions prior to each storm.

Many of the factors that influence erosion, particularly soil, slope and land use, have considerable spatial variability and cannot be described by a single average value, even over areas as small as one field. Spatially lumped models, which treat an area as a single unit of uniform characteristics, are not appropriate for most natural catchments. If this spatial variability is to be taken into account, a dynamic distributed model must be used. The simulation of erosion contained in EUROSEM is linked to the water and sediment routing structure of the KINEROS model (Woolhiser *et al.*, 1990). It has also been linked with the MIKE SHE model (Danish

 

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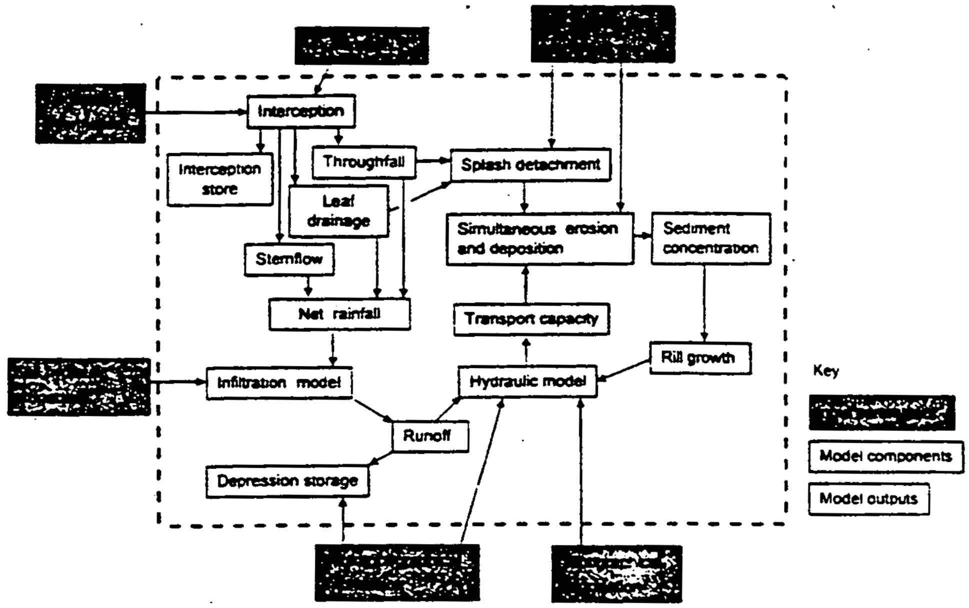


Figure 1. Flow chart of EUROSEM

Hydraulic Institute, 1993) which is a continuous simulation model and an extension of the original Système Hydrologique Européen (SHE) model (Danish Hydraulic Institute, 1985; Abbott *et al.*, 1986). The description given in this paper is for the EUROSEM–KINEROS version.

## BASIC CONCEPTS

The flow chart for EUROSEM is presented in Figure 1. EUROSEM has a modular structure with each module being developed in as much detail as the existing level of knowledge permits. This structure will enable continuous improvements to be made in the light of new research. EUROSEM uses a dynamic and distributed simulation of erosion processes with explicit simulation of rill and interill flow. Water and sediment are routed over the land surface which is represented, following KINEROS, as a series of interlinked uniform slope planes and channel elements to simulate small catchments. Soil loss is computed as a sediment discharge, defined as the product of the rate of runoff and the sediment concentration in the flow, to give a volume (or mass) of sediment passing a given point in a given time. The computation is based on a numerical solution of the dynamic mass balance equation (Bennett, 1974; Kirkby, 1980; Woolhiser *et al.*, 1990):

$$\frac{\partial(AC)}{\partial t}+\frac{\partial(QC)}{\partial x}-e(x,t)=q_s(x,t) \tag{1}$$

where $C$=sediment concentration ($m^3\ m^{-3}$), $A$=cross-sectional area of the flow ($m^2$), $Q$=discharge ($m^3\ s^{-1}$), $q_s$=external input or extraction of sediment per unit length of flow ($m^3 s^{-1} m^{-1}$), $e$=net detachment rate or rate of erosion of the bed per unit length of flow ($m^3\ s^{-1}\ m^{-1}$), $x$=horizontal distance (m), and $t$=time (s).

For channel flow $q_s$ represents lateral inflows of sediment from the base of adjacent hillsides. When applied to overland flow over hillslopes, $q_s$ becomes zero.

The term $e$ in Equation 1 is composed of two major components:

$$e = DR + DF \tag{2}$$

where $DR$=the rate of soil particle detachment by raindrop impact ($m^3\ s^{-1}\ m^{-1}$), and $DF$=the net rate of soil particle detachment by the flow (positive for detachment and negative for deposition) ($m^3\ s^{-1}\ m^{-1}$).




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Values of surface runoff $Q(x,t)$ and $A(x,t)$ are obtained by numerical solution of the dynamic mass balance equation for water, analogous to Equation 1:

$$\frac{\partial A}{\partial t}+\frac{\partial Q}{\partial x}=w[r_i(t)-f(t)] \qquad (3)$$

where $r_i(t)$ = the rainfall rate less the interception (mm min$^{-1}$), $f(t)$ = the local infiltration rate (mm min$^{-1}$) and $w$ = flow width (mm).

Equation 3 is solved using a kinematic wave assumption for a fixed relation $Q(A)$ (Woolhiser and Liggett, 1967; Woolhiser, 1969). During flow recession, the calculations of infiltration loss, $f(t)$, following KINEROS, are based on the depth of water ($h(x,t)$, m) and an estimate of the fraction of the surface covered to this depth.

## MODEL COMPONENTS

### *Rainfall interception*

Rainfall input to the model is in the form of break-point data giving a depth $R$ (mm) for each time step during a storm. From this input, rainfall intensity ($R_i$, mm h$^{-1}$) and rainfall volume (m$^3$) are calculated. Account is also kept of the cumulative rainfall ($R_{cum}$, mm) received during the storm.

On reaching the canopy of the vegetation, the rainfall is divided into two parts, namely that falling either on open ground or passing through gaps in the canopy and reaching the soil surface as direct throughfall ($DT$, mm), and that which strikes the vegetation cover. The division is based on the simple relationship:

$$IC=R\cdot COV \qquad (4)$$

where $IC$ = the depth of rainfall intercepted by the vegetation (mm), and $COV$ = the fraction of the surface covered with vegetation.

An initial proportion of the intercepted rainfall is stored on the leaves and branches of the vegetation. This is termed the interception store. The rainfall held in this store does not reach the soil surface and therefore is unavailable for infiltration or runoff. In many erosion models, this interception store is either ignored, as in CREAMS, or is considered as a depth which has to be filled before rain is allowed to pass from the vegetation canopy to the ground, as in KINEROS. This last approach means that no rain reaches the soil surface from the canopy until the interception store is full. EUROSEM adopts a more dynamic approach which allows rainfall to pass from the canopy to the ground at the same time as the interception store is being filled. This means that some transfer of water from the canopy will take place right from the start of the storm. The depth of the interception store ($IC_{store}$, mm) for a time step ($t_s$, s) is modelled as a function of the cumulative rainfall from the start of the storm, using the exponential relationship proposed by Merriam (1973):

$$IC_{store}=IC_{max}[1-\exp(R_{cum}/IC_{max})] \qquad (5)$$

where $IC_{max}$ = the maximum depth of the interception store for the given crop or vegetation cover (mm).

In most erosion models, it is assumed that the canopy cover protects the underlying soil from erosion and so little attention is given to the fate of intercepted rainfall and its influence on the erosion system. In EUROSEM, however, the importance of leaf drips in detaching soil particles from the soil mass is recognized, so it is necessary to calculate the proportion of the rainfall reaching the ground surface as leaf drainage. Thus, the rainfall which is intercepted by the canopy and not held in the interception store (termed temporarily intercepted throughfall ($TIF$, mm)) is partitioned into stemflow ($SF$, mm) and leaf drainage ($LD$, mm). The depth of stemflow for each time step is modelled as a function of the average acute angle ($PA$, degrees) of the plant stems to the ground surface, using equations developed in laboratory experiments by van Elewijck (1989a,b). These

 

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equations have been modified by assuming that a maximum of half the depth of temporarily intercepted throughfall is available for stemflow, to give:

$$SF = 0{\cdot}5\, TIF\, (\cos PA \sin^2 PA) \tag{6}$$

for grasses, and

$$SF = 0{\cdot}5\, TIF \cos PA \tag{7}$$

for other plant species.

Conceptually, Equations 6 and 7 describe the relationship between the diameter of the catching surface (stems and leaves) and the median volume drop diameter of the raindrops. Where, as with grasses, the mean diameter of the catching surface is less than the drop diameter, gravity, expressed by $\sin PA$, plays an important role in determining the depth of stemflow. With thicker catching surfaces, stemflow depth depends only on the projected length of the stems or leaves, as expressed by $\cos PA$.

The difference between the depth of the temporarily intercepted throughfall and the depth of stemflow for each time step comprises leaf drainage, i.e. that component of the rainfall which reaches the soil surface as individual drips from the leaves. The net rainfall at the ground surface, which is therefore available for infiltration, is the summation of the direct throughfall, stemflow and leaf drainage.

*Infiltration*

KINEROS–EUROSEM models infiltration using the equation (Smith and Parlange, 1978):

$$f_c = K_s \frac{\exp(F/B)}{\exp(F/B) - 1} \tag{8}$$

where $f_c$ = the maximum rate at which water can enter the soil, which is known as the infiltration capacity (mm min$^{-1}$), $K_s$ = the saturated hydraulic conductivity of the soil (mm min$^{-1}$), $F$ = the amount of rain already absorbed by the soil (mm), and $B$ = an integral capillary and water deficit parameter (mm) related to soil fraction <2 mm.

The term $B$ is obtained from:

$$B = G(\theta_s - \theta_i) \tag{9}$$

where $G$ = the effective net capillary drive (mm), $\theta_s$ = the maximum value of water content of the soil (m$^3$ m$^{-3}$), and $\theta_i$ = the initial value of soil water content (m$^3$ m$^{-3}$).

The term $G$ is a conductivity-weighted integral of the capillary head of the soil, used in many infiltration equations, and defined as:

$$G = \frac{1}{K_s} \int_{-\infty}^{0} K(\psi)\, d\psi \tag{10}$$

where $\psi$ = the soil matric potential (mm), and $K(\psi)$ = a hydraulic conductivity function.

$G$ is essentially a property of the soil with units of length and is conceptually equivalent to a value of effective capillary head. The parameters $G$, $K_s$ and $\theta_s$ are measurable and physically related, but guidelines for their estimation based on soil texture are given by Woolhiser *et al.* (1990).

In the initial part of a storm, $F$ is increased by the addition of rainfall, since the value of $f_c$ is very large for small values of $F$. The model predicts the initiation of runoff as the time when $F$ grows to the point that Equation 8 finds $f_c$ to be equal to or less than the rain rate ($r_i$), beyond which $f(t) = f_c\,(F)$. The prediction of ponding and

 

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subsequent infiltration rate in a unified equation is based on soil physics, and is the main reason behind using $F$ rather than $t$ as the independent variable (Smith, 1983).

When operating EUROSEM, the input values of $G$, $K_s$ and $\theta_s$ are varied according to the soil texture, degree of crusting and management practice. Adjustments to these values are made within the model to account for the effects of rock fragments and the presence of any vegetation or crop cover. Rock fragments affect infiltration in two ways. First, they reduce the effective overall storage in porosity $(\theta_s - \theta_i)$ by modifying the parameter $B$ in Equation 8 to account for the presence of rock fragments ($ROC$) using the relationship (Woolhiser *et al.*, 1990):

$$B_{roc} = B(1 - ROC) \tag{11}$$

where $B_{roc}$ = the parameter $B$ modified for rock fragments (mm min$^{-1}$), and $ROC$ = the fraction of the soil composed of rock fragments, expressed as a volume between 0 and 1.

Second, rock fragments affect infiltration into soils through their position on the soil surface (Poesen and Ingelmo-Sanchez, 1992; Poesen *et al.*, 1994). Rocks which are embedded in a surface seal reduce infiltration whereas rocks which sit on the surface or are surrounded by macropores (e.g. those produced as a result of tillage) protect the soil structure and promote infiltration. EUROSEM models the first condition using the equation:

$$K_{sroc} = K_s(1 - PAVE) \tag{12}$$

and the second using the equation:

$$K_{sroc} = K_s(1 + PAVE) \tag{13}$$

where $K_{sroc}$ = a modified value of saturated hydraulic conductivity (mm min$^{-1}$), and $PAVE$ = the fraction of the surface area covered by rock fragments (between 0 and 1).

Equation 13 should not be used, however, if the effects of tillage on saturated hydraulic conductivity outweigh the effects of rock fragments.

The research base for modelling the effect of vegetation cover on infiltration is rather sparse. Thornes (1990) proposes that infiltration capacity increases exponentially with increasing percentage vegetation cover as a function of increases in organic matter and decreases in the bulk density of the soil. Such a relationship is similar to that developed by Holtan (1961) to express the saturated hydraulic conductivity of the soil as a function of the percentage basal area of the vegetation. Based on his work, the following equation is used in EUROSEM to modify the saturated hydraulic conductivity value of the soil:

$$K_{sveg} = K_s \frac{1}{1 - PBASE} \tag{14}$$

where $K_{sveg}$ = the saturated hydraulic conductivity of the soil with the vegetation (mm min$^{-1}$), $PBASE$ = the total area of the base of the plant stems expressed as a proportion (between 0 and 1) of the total area of the plane.

*Soil surface condition*

The soil surface is considered to be composed of form roughness and hydraulic roughness, both of which play a role in erosion as well as water flow. The form roughness of the soil surface determines the volume of water than can be held on the surface as depression storage. The basis for modelling, however, is extremely limited and depression storage is ignored in many hydrological models. However, it is included in EUROSEM where it can be used to describe different surfaces produced by tillage. The form roughness of the soil surface is expressed by a roughness measure ($RFR$), defined as the ratio of the straight line distance between two points on the ground ($X$, m) to the actual distance measured over all the microtopographic irregularities ($Y$, m). $RFR$ is




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determined in the field by placing a one metre long chain with a 5 mm link on the soil surface and measuring the distance between the two ends of the chain:

$$RFR = \frac{Y - X}{Y} \times 100 \quad (15)$$

This measure is converted into a surface storage depth, $D$(m), using a regression equation developed from research in southern Germany:

$$D = \exp(-6{\cdot}6 + 0{\cdot}27\, RFR) \quad (16)$$

*Surface runoff processes*

When the net rainfall intensity at the ground surface exceeds the infiltration rate and surface depression storage is satisfied, the excess comprises surface runoff. In KINEROS–EUROSEM, runoff along a slope for a plane element, a rill, or a channel is viewed as a one-dimensional surface flux in which discharge ($Q$) is related to the hydraulic radius ($r$). The rating equation, based on the normal flow equation, is:

$$u = \alpha\, r^{m-1} \quad (17)$$

where, based on the Manning equation for flow velocity, $u$=flow velocity (m s$^{-1}$), $r$=hydraulic radius (m), $\alpha = s^{0.5}/n$ (m$^{1/3}$ s$^{-1}$), $s$=slope (%), $n$=Manning roughness value (m$^{-1/3}$ s), and $m$=5/3.

Assuming discharge $Q = uA$, the general rating equation can be written as:

$$Q = \alpha\, p\, r^{m} \quad (18)$$

where $p$=the wetted perimeter (m).

Combining Equation 18 with the continuity Equation 3 gives:

$$\frac{\partial A}{\partial t} + \alpha\left[ m p r^{m-1} \frac{\partial r}{\partial x} + r^{m} \frac{\partial p}{\partial x} \right] = w q(x, t) \quad (19)$$

where $q = r_i - f$ is the lateral inflow rate (m$^3$ s$^{-1}$ m$^{-1}$), or 'rainfall excess'.

Manning's equation for flow velocity is chosen because of its wide use by engineers and availability of input data. It is recognized that this implies turbulent flow but, given the types of storms for which EUROSEM is designed, this does not seem unreasonable. Also, it avoids the need to identify the transition between early laminar and turbulent flow when, as is generally the case, the flow is disturbed by raindrop impacts.

The traditional concepts of rill and interrill processes, where flow erosion occurs in the rills and splash erosion on the interrill area, are not adopted by EUROSEM. Instead, splash and flow processes are modelled on all areas, with the distinction between rill and interrill areas being one of geometry. Rills, which can encompass any form of concentrated flow path, are described as essentially trapezoidal channels with side wall slopes of 0 (vertical) or greater (Figure 2). Interrill areas are surfaces without orientated roughness. The whole area may or may not be rilled, but if rills are present EUROSEM assumes the interrill areas slope towards the rills rather than straight downslope. Erosion and deposition by splash and flow detachment can occur at any point on either the rill or interrill areas (see next two sections). Water and sediment are routed across the interrill areas as well as through any rills until the bottom of the plane is reached.

Within EUROSEM it is possible to specify whether the rills are uniform in depth over the whole length of a plane or whether their depth increases downslope. The depth and width of the rill at each point along its length will change in response to erosion and deposition. EUROSEM assumes that deposition will reduce the depth

 

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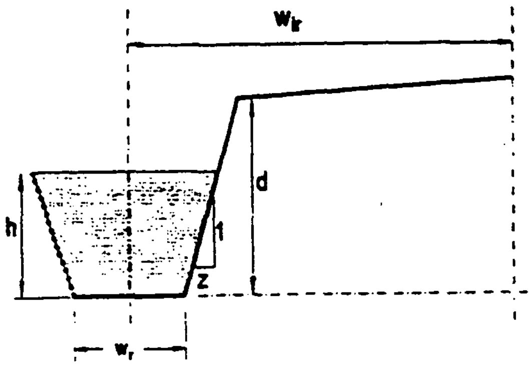


Figure 2. Representation of the hydraulic geometry of a rill channel ($W_{ir}$=distance between the centre line of the rill and centre line of the interrill area; $d$=depth of base of the rill below the average height of the interill surface; $z$=side slope of the rill expressed as the ratio of horizontal to vertical component)

whereas erosion will increase the width and the depth by taking material equally from the two sides and bottom of the rill.

For shallow interrill surface flow, a unit width is used for computations, so that $p=1$ and $r=h$. In EUROSEM, $h$ is a lumped variable representing the mean of a distribution of depths across the slope. The discharge rating Equation 18 becomes:

$$Q = a\,h^m \tag{20}$$

which represents the mean (non-linear) condition for such an averaged concept of $h$. Equation 19, for a unit width ($p,w=1$) and $r=h$, becomes:

$$\frac{\partial h}{\partial t} + \alpha m h^{m-1}\frac{\partial h}{\partial x} = q(x,t) \tag{21}$$

In KINEROS, the kinematic wave Equations 19 or 21 are solved numerically for a finite difference grid by a four-point implicit method using the Newton–Raphson technique (Pearson, 1983; Woolhiser *et al.*, 1990). The upslope boundary condition for the depth of flow ($h$) at $x=0$ and $t=0$ is either 0 or equal to the depth of runoff from an upslope contributing plane. The finite difference solution of Equation 1 provides arrays of nodal point values of $Q$, $A$ and $u$. These values, along with the array of nodal values of $C$ at the previous time step ($C_i$, $i=1$, N) (plus the upstream and initial conditions described below), allow an explicit solution of the finite difference form of Equation 1 for new values of $C_i$, starting with the node just below the upstream boundary.

A similar procedure is adopted for routing flow in rills or channels, where the relevant rating equation is Equation 19. The term $q(x,t)$ in the equation then becomes the unit discharge into the rills from interrill contributions.

*Soil detachment by raindrop impact*

Modelling this process is based on relationships between detachment and the kinetic energy of rainfall. In contrast with most erosion models which, as seen earlier, ignore the effects of leaf drip, EUROSEM explicitly models soil detachment by raindrop impact for both direct throughfall and leaf drainage as a function of their kinetic energy. This enables the effects of different heights of vegetation and canopy and residue covers to be simulated explicitly.

The rainfall energy reaching the ground surface as direct throughfall ($KE(DT)$, J m$^{-2}$ mm$^{-1}$) is assumed to be the same as that of the natural rainfall. It is estimated as a function of rainfall intensity from the equation derived

 

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by Brandt (1989) which assumes that the raindrop size distribution follows that described by Marshall and Palmer (1948):

$$KE(DT) = 8\cdot 95 + (8\cdot 44 \log R_i) \tag{22}$$

The energy of the leaf drainage ($KE(LD)$, $\mathrm{J\,m^{-2}\,mm^{-1}}$) is estimated from the following relationship developed experimentally by Brandt (1990):

$$KE(LD) = (15\cdot 8PH^{0\cdot 5}) - 5\cdot 87 \tag{23}$$

where $PH$ = the effective height of the plant canopy (m).

The relationship is considered valid because, for a wide range of plants, the drop-size distribution of leaf drainage (with a median drop diameter of about 4·8 mm) has been found not to vary (Brandt, 1989). This means that the variations in the energy of leaf drainage are solely a function of the impact velocity of the raindrops which depends on the height of fall. The model sets the kinetic energy of leaf drainage to zero when the canopy height is less than 14 cm to avoid the negative values predicted by Equation 23.

The total kinetic energy of the rainfall can be calculated by multiplying the energies obtained from Equations 22 and 23 by the respective depths of direct throughfall and leaf drainage received and summing the two values. This calculation is made in EUROSEM for every time increment of the rainstorm.

*Detachment equation for rainfall.* Soil detachment by raindrop impact ($DR$, $\mathrm{m^3\,s^{-1}\,m^{-1}}$) for time step ($t_s$) is calculated from the equation:

$$DR = \frac{k}{\rho_s} KE\, e^{-zh} \tag{24}$$

where $k$ = an index of the detachability of the soil ($\mathrm{g\,J^{-1}}$) for which values must be obtained experimentally, $\rho_s$ = particle density ($\mathrm{kg\,m^{-3}}$), $KE$ = the total kinetic energy of the net rainfall at the ground surface ($\mathrm{J\,m^{-2}}$), $z$ = an exponent varying between 0·9 and 3·1, depending on soil texture but for which a value of 2·0 can be used for a wide range of conditions (Torri *et al.*, 1987), and $h$ = the mean depth of the surface water layer (m).

The influence of slope on soil particle detachment is neglected in EUROSEM because of the difficulty of characterizing the 'effective slope' which needs to be measured over distances of several drop diameters from the point of raindrop impact and is dependent upon local surface roughness and the angle of internal friction of the soil (Torri and Poesen, 1992). It is not the same as the general surface slope, which is generally smaller.

Where non-erodible surfaces, such as rock outcrops, surface rock fragments, concrete and tarmac, occur within the element, the detachment rate is modified by:

$$DR_{pav} = DR(1 - PAVE) \tag{25}$$

where $DR_{pav}$ = the detachment rate allowing for the non-erodible surfaces ($\mathrm{m^3\,s^{-1}\,m^{-1}}$), and $PAVE$ = the fraction (between 0 and 1) of the soil surface covered by non-erodible surfaces.

*Initial condition for sediment concentration.* Since, during a rainstorm, splash erosion will already be taking place when runoff begins, the initial sediment concentration in the runoff cannot be taken as zero. Based on an analysis of Equation 1 at the time of ponding ($t_p$, s) or $x=0$ and $A=0$, the sediment concentration ($C$) at $t_p$ is calculated, as in KINEROS, from:

$$C(t_p) = \frac{DR}{q + v_s} \tag{26}$$

where $v_s$ is the particle settling velocity ($\mathrm{m\,s^{-1}}$).




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This equation is also used to determine the boundary condition at the upper end of a slope plane when there is no input of runoff or sediment from above.

*Soil detachment by runoff*

Soil detachment by runoff is modelled in terms of a generalized erosion–deposition theory proposed by Smith *et al.* (1995). This assumes that the transport capacity concentration of the runoff ($TC$, $m^3\ m^{-3}$) reflects a balance between the two continuous counteracting processes of erosion and deposition. It implies that the ability of flowing water to erode its bed is independent of the amount of material it carries and is only a function of the energy expended by the flow, particularly the shear between the water and the bed, and the turbulent energy in the water. The erosion rate of the flow ($E_q$, $m^3\ s^{-1}\ m^{-1}$) is continually accompanied by deposition at a rate equal to $w\, C\, v_s$, where $w$ is the width of flow (m). This condition can be expressed as:

$$DF = E_q - w\, C\, v_s \tag{27}$$

where $DF$ = the net detachment rate of soil particles by the flow (Equation 2), which is positive for erosion and negative for deposition.

According to this generalized theory, the transport capacity concentration ($TC$) represents the sediment concentration at which the rate of erosion by the flow and accompanying rate of deposition are in balance. In this condition, $DF$ is zero and $E_q$ equates to the deposition rate ($wTC\, v_s$). A general equation for net soil detachment during flow, expressed in terms of settling velocity and transport capacity, then becomes:

$$DF = w\, v_s\, (TC - C) \tag{28}$$

This equation, however, assumes that the soil particles are loose so that processes are reversible whereas, in reality, detachment will be limited by factors such as soil cohesion. The net pick-up rate for cohesive soil therefore needs to be reduced by a coefficient whenever $C$ is less than $TC$. Equation 28 becomes:

$$DF = \beta\, w\, v_s\, (TC - C) \tag{29}$$

where $\beta$ = a flow detachment efficiency coefficient. This coefficient is equivalent to the efficiency functions proposed by Rose *et al.* (1983) and Styczen and Nielsen (1989) in their procedures for modelling of soil detachment by flow.

By definition, $\beta = 1$ when $DF$ is negative (deposition is occurring) and $\beta < 1$ for cohesive soils when $DF$ is positive ($TC > C$). This parameter can be evaluated experimentally by a steady flow experiment in which clear water flows across a uniform bed of soil, and the spatial rate of change of $C$ is measured. Such experiments have in fact been done in furrow irrigation (Trout, 1996). EUROSEM estimates $\beta$ as a function of the cohesion of the soil ($J$, kPa) as measured by a torvane under saturated conditions. Soil cohesion has been successfully related to the onset of rilling (Rauws and Govers, 1988), and is recognized by a number of authors (Torri and Borselli, 1991; Brunori *et al.*, 1989; Crouch and Novruzi, 1989) as related to erodibility. It is recognized, however, that the relationship between soil cohesion and detachability of the soil by runoff is dependent upon the initial moisture content (Govers and Loch, 1993) and on initial structural conditions (Govers *et al.*, 1990), and that no unique relationship exists even for a single soil. However, until a procedure is developed that enables the detachability of soils by flow to be predicted from soil parameters which can be easily measured in the field, an approach based on cohesion seems the most appropriate. For application, EUROSEM assumes at present that when $J < 1$ kPa, $\beta = 1$. For larger values of $J$, the value of $\beta$ is reduced exponentially:




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$$\beta = 0{\cdot}79\,e^{-0{\cdot}85J} \tag{30}$$

When $TC$ is zero and $DR$ has a value due to rainfall energy, a value of $C$ will be obtained such that, using Equation 2 with $e=0$, $DET=w\,v_s\,C$. The concentration in flow will be $C=DET/w\,v_s$. This has been termed 'rain flow transportation' (Moss *et al.*, 1979).

*Transport capacity of the flow*

The capacity of runoff to transport detached soil particles is expressed in EUROSEM in terms of a concentration, $TC$. For flow in rills, it is modelled as a function of unit stream power, using a relationship based on the work of Govers (1990) which showed that the transporting capacity of overland flow could be predicted from simple hydraulic parameters. For interrill flow, $TC$ is modelled as a function of modified stream power, based on the experimental work of Everaert (1991).

*Rill transport capacity.* Unit stream power ($\omega$, $\mathrm{cm\,s^{-1}}$) is the hydraulic variable on which rill $TC$ is based, and is defined as:

$$\omega = 10\,u\,s \tag{31}$$

where $s$=slope (%), and $u$=mean flow velocity ($\mathrm{m\,s^{-1}}$).

Based on this variable, Govers (1990) found that $TC$ could be expressed for any particle size (ranging from 50 to 250 $\mu$m) as follows:

$$TC = c(\omega - \omega_{cr})^{\eta} \tag{32}$$

where $\omega_{cr}$=critical value of unit stream power (=$0{\cdot}4\,\mathrm{cm\,s^{-1}}$), and $c$,$\eta$=experimentally derived coefficients depending on particle size.

Further analysis has shown that one can estimate $c$ and $\eta$ as follows:

$$c = [(d_{50}+5)/0{\cdot}32]^{-0{\cdot}6} \tag{33}$$

$$\eta = [(d_{50}+5)/300]^{0{\cdot}25} \tag{34}$$

where $d_{50}$ is the median particle size of the soil ($\mu$m).

These relationships were derived from over 500 observations in experiments carried out on a range of materials with median grain sizes ($d_{50}$) from silt to coarse sand, slopes from 1 to 15 per cent and discharges from 2 to 100 $\mathrm{cm^3\,cm^{-1}\,s^{-1}}$. They are valid for sediment concentrations up to 0·32 which seemed to be an upper limit obtained in the experiments beyond which further increases in stream power caused no further increase in sediment concentration. The need to insert a critical value for unit stream power of 0·4 $\mathrm{cm\,s^{-1}}$ means that the equations cannot be used at very low unit stream powers.

*Interrill transport capacity.* EUROSEM uses the following interrill flow equations based on experimental work on shallow interrill flow by Everaert (1991), using a range of particle sizes from 33 to 390 $\mu$m:

$$TC = \frac{b}{\rho_s q}\left[(\Omega - \Omega_c)^{0{\cdot}7/n} - 1\right]^{\kappa} \tag{35}$$




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where $\kappa=5$, $\rho_s$ = the sediment density ($kg\,m^{-3}$), and b = a function of particle size defined by:

$$b=\frac{19-(d_{50}/30)}{10^4} \quad (36)$$

$\Omega$ = the modified stream power ($g^{1.5}\,cm^{-2/3}\,s^{-4.5}$) defined by Bagnold as:

$$\Omega=\frac{\left(U^{*}u\right)^{3/2}}{h^{2/3}} \quad (37)$$

in which $U^*$ = shear velocity ($m\,s^{-1}$). The symbol $\Omega_c$ is a critical value of $\Omega$ found by using, in the same formula, the Shields critical shear velocity (White, 1970):

$$U_c^{*}=\sqrt{y_c(\rho_s-1)gd_{50}} \quad (38)$$

where $y_c$ is the modified Shields' critical shear velocity based on particle Reynolds number ($m\,s^{-1}$), and $g$ is the acceleration due to gravity ($m\,s^{-2}$).

## CALCULATION OF SOIL EROSION

### *Hillslope erosion*

For each time step and each node along the slope plane, the net rate of erosion ($e$) and the sediment discharge (product $QC$) are calculated. Combining Equations 2 and 29, $e$ is obtained as:

$$e=DR+\beta w\,v_s(TC-C) \quad (39)$$

When rates of soil detachment by raindrop impact are sufficiently small and the sediment concentration in the flow exceeds the transport capacity, $e$ becomes negative and represents a net deposition rate. This situation arises when $DR$ is very low or when runoff and sediment are routed from one slope plane to another of lower gradient.

Three cases of surface topography can be simulated, based on the geometry of the rill and interrill areas (see section on 'surface runoff processes'):

(a) the surface may contain no rills, but have some surface irregularities;
(b) the surface may be rilled, with interrill flows routed toward the rills as described by Equation 21;
(c) the surface may be furrowed, or have very dense rills, such that interrill routing is illogical due to the short distance traversed by interrill flows.

For case (a), typical of relatively smooth slope planes, interrill flow is assumed over the entire element, and the flow direction is directly down the plane. Interrill splash and transport relations are used. For case (b), the model simulates both shallow flow and rainsplash erosion between the rills, and downslope flow with much large carrying capacities in the rills. Interrill flow is routed towards the rills along a slope taken as the vector sum of the slope along the rills and the slope of the surface in a direction normal to the rills; rill spacing on the element is assumed to be uniform. When distance of interrill flow is less than 1 m, as in case (c), interrill routing is inappropriate, and rill input rate $q$ is taken as the rainfall excess rate times the interrill flow distance. Rain flow transport concentrations are used for interrill sediment concentrations. By using case (c), EUROSEM can simulate the furrows produced by agricultural implements and also plough-rill erosion.

### *Channel erosion*

Channel flow and erosion are simulated in EUROSEM using the same general approach adopted for rill erosion. The main differences are that soil detachment by raindrop impact within the channel is neglected and that lateral inflows of sediment from the hillsides ($q_s$ in Equation 1) become important. In the same manner as

 

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for rills, $C$ may be explicitly computed for each time step in the finite difference scheme, once the array of hydraulic variables, $A$, $Q$ and $u$, is found, starting with the first node below the upper boundary. If there is no input of runoff at the upper end of the channel, the transport capacity at the first node is zero and the boundary condition is set as:

$$C(0,t) = \frac{q_s}{Q + v_s BW} \tag{40}$$

where $BW$=the bottom width of the channel (m). Otherwise, procedures are precisely the same as for calculation of rill sediment transport. Bank collapse is not simulated.

*Catchment representation*

As in the KINEROS model, a catchment is represented as a network of surfaces and channels of rather arbitrary complexity. Channels may receive distributed inputs from hillslopes on either or both sides or as a concentrated flow from upstream (as for a headwater area). Channels may also receive input from one or two upstream channels. Hillslopes may be represented as heterogeneous along their flowpaths in slope, width, rill density or other properties by using a cascade of adjacent surfaces (Morgan *et al.*, in press).

## MODEL APPLICATIONS

We acknowledgte that a validation of this model is beyond the scope of an introductory paper, and this section is rather intended to demonstrate briefly some of the unique capabilities of EUROSEM and dynamic erosion models in general. More extensive field applications are intended for subsequent publications. Data collected from the Woburn Erosion Reference Experiment (Catt *et al.*, 1994), a series of erosion plots operated jointly by Silsoe College and Rothamsted Experimental Station at Woburn, Bedfordshire, UK, are used for a simulation to demonstrate the output that can be obtained from EUROSEM (Version 3.4). The experiment is sited on a sandy loam soil of the Cottenham Series, a brown sand, as defined by Avery (1980), classified as an Udipsamment (Soil Survey Staff, 1975) and a Cambic Arenosol (FAO–UNESCO, 1974), developed on Cretaceous Lower Greensand (Catt *et al.*, 1974). The plot used for the simulation is 40 m long and 22·5 m wide with a mean slope of 9 per cent. Hydrographs and sedigraphs were recorded using the sediment sampling system described by Vivian and Quinton (1993). To take account of under-recording of the pumped samplers, the sediment concentrations were corrected by multiplying them by the ratio of the total soil loss determined in the field to that determined from the pumped samples. The event chosen for simulation, a storm of 9·8 mm on 17 September 1992, is a characteristic short-duration, high-intensity summer storm for southern England. At the time of the storm, the plot was under a crop of sugar beet with a 70 per cent cover. The main pulse of the storm lasted 20 min with a peak intensity of 126 mm $h^{-1}$ for 1 min (Figure 3).

The output hydrograph and sediment graph for the storm, together with observed data, are illustrated in Figure 3. A best fit for the hydrograph was obtained by a trial-and-error calibration, with all parameters constrained to physically realistic values. EUROSEM appears to simulate quite well the time to peak discharge, the peak discharge and the overall shape of the hydrograph. The sediment graph, for which no calibration was attempted, shows that EUROSEM appears to overestimate the sampled sediment concentrations, but the differences are not large.

EUROSEM can also provide information on changes in surface elevation and rill geometry, as shown in Figure 4 where it can be seen that the greatest changes have taken place at the lower end of the plot where discharge is higher. Figure 5 illustrates the dynamic modelling of the interception process within EUROSEM. Under the sugar beet canopy on the plot, most of the water intercepted by the plant canopy reaches the ground as leaf drip with the rest being made up of stemflow and direct throughfall; only a relatively small amount is held in the interception store. Comparing Figure 5 with the rainfall hyetograph in Figure 3, the relative importance of leaf drip is seen to increase with rainfall intensity.

The dynamics of surface water flow during a storm of rapidly varying rainrates are important in erosion and deposition properties. For most field slope lengths, storms rarely, if ever, reach steady flow, as assumed by




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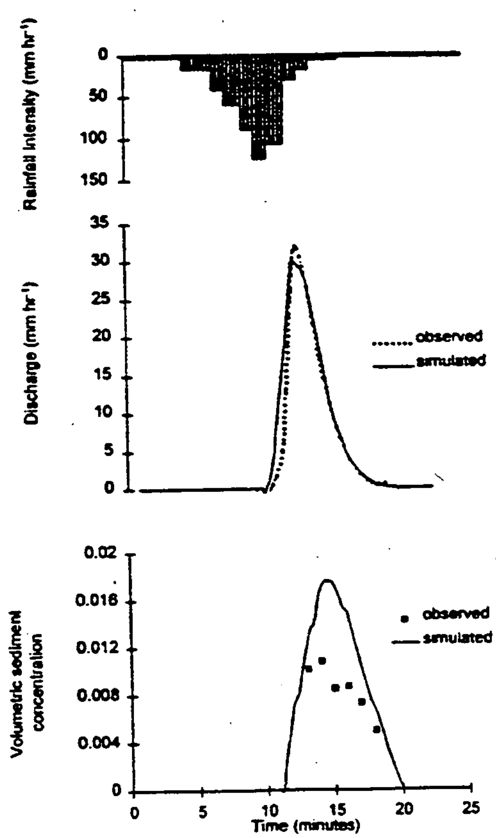


Figure 3. Hyetograph, simulated and observed hydrographs, and simulated and observed sediment graphs for the event of 17 September 1992. (The simulated hydrograph was fitted to the observed data. Although the simulated sediment concentrations appear too high, the simulated total storm soil loss was 57 kg compared with an observed value of 42 kg, suggesting that the measured sediment concentrations are in error.)

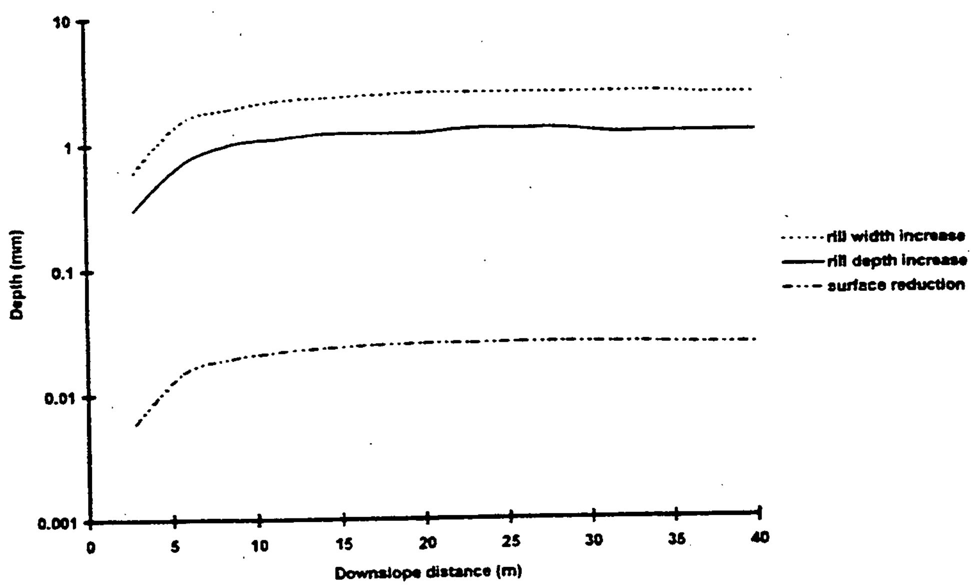


Figure 4. Simulated changes in rill geometry and surface elevation with distance downslope





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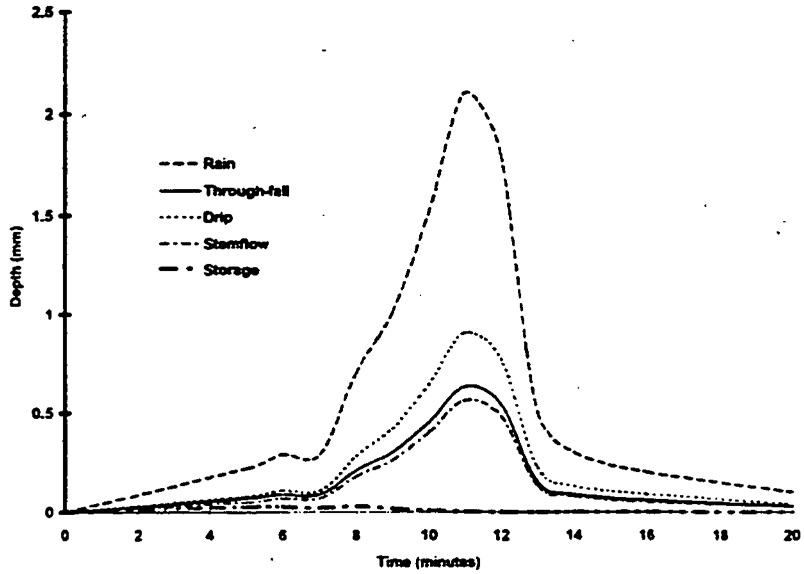


Figure 5. Simulation of the division of rainfall into leaf drip, direct throughfall, stemflow and interception storage

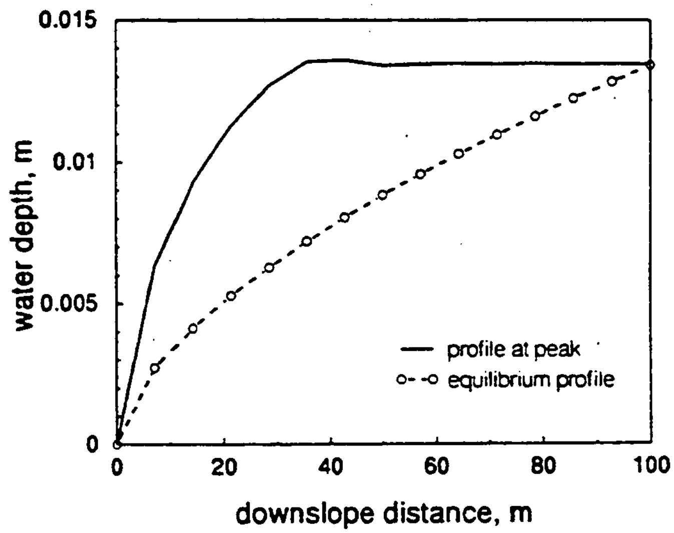


Figure 6. Comparison of the actual water profile with the steady flow profile at peak flow for the storm of Figure 3 on a 100m slope

several sediment yield models. For the storm of Figure 3, for example, on a 100 m slope, the actual surface water profile at time of flow peak is compared with a steady flow profile in Figure 6. This is a simple slope shape, but it is clear that the steady flow assumption cannot actually predict with much accuracy where erosion and deposition will occur. Since the WEPP methodology finds the peak flow using kinematic water routing, the accuracy of that method will depend critically on the extent to which the sediment concentration at estimated peak flow approximates the overall average $C$. Further, as shown by Smith *et al.* (1995), steady flow assumptions which lie behind the slope length extrapolations in the USLE and RUSLE approaches are considerably in error for longer slopes, for the same reason: steady flow does not occur under erosive storms on longer slopes.

## CONCLUSIONS

EUROSEM is a fully dynamic state-of-the-art erosion model, able to simulate sediment transport, erosion and deposition by rill and interrill processes over the land surface in individual storms for both single fields and




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small catchments. It provides information on total runoff, total soil loss, the storm hydrograph and storm sediment graph. At present, EUROSEM is not able to simulate erosion by ephemeral gullies or by saturation overland flow. Compared with other models, however, EUROSEM provides for explicit consideration of rill and interrill flow, improved simulation of plant cover effects on interception, rainfall energy and flow velocity, transport capacity computations using relationships based on over 500 experimental observations made with shallow surface flows, and specific simulation of the effects of rock fragments (stoniness) on infiltration, flow velocity and splash erosion. The true dynamic nature of the simulation provides advantages over approximate steady flow methods now in use.

Readers interested in obtaining a copy of EUROSEM may visit http://www.silsoe.cranfield.ac.uk/eurosem/eurosem.htm

## ACKNOWLEDGEMENTS

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# Grundvandets sårbarhed - EDB-modellering

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## Indledning og problemstilling

Debatten omkring beskyttelsen af grundvandet pågår med uformindsket styrke. Begreber som kildepladszone, indvindingsoplande, infiltrationsområder, beskyttelseszoner og særlige drikkevandsområder svirrer i luften. Der er lagt op til anvendelse af endog betydelige ressourcer i forbindelse med f.eks. tvungne ændringer af areal-anvendelse og i forbindelse med udlægning af beskyttelseszoner for grundvandsindvindinger. Der må derfor også ofres ressourcer på at bestemme og prioritere disse områder bedst muligt.

Udlægning af områder, hvor der pålægges visse restriktioner på arealanvendelsen, f.eks. mindsket brug af pesticider og gødning, for at sikre den fremtidige kvalitet på det nedsivende grundvand, vil formentlig blive mere og mere almindeligt. Man er allerede startet i Nordjyllands Amt med egentlig opkøb af landbrugsarealer og efterfølgende udlægning til skovbrug. Der vil derfor blive brug for troværdige metoder til udpegning af de områder, hvor grundvandet egentlig dannes, og formentlig også for at vide, hvor lang tid der går fra grundvandet dannes, til det genfindes i indvindingsboringerne.

Indvindingsoplandet til en given indvindingsboring eller gruppe af indvindingsboringer (kildeplads) kan defineres som det område i den filtersatte formation indenfor hvilket en tilstedeværende vandpartikel før eller siden vil nå hen til indvindingsboringen, se Figur 1a. Dvs. dette område bestemmes alene på grundlag af strømningsforholdene i det magasin, hvor boringen er filtersat. Dette område fokuseres der fra for-



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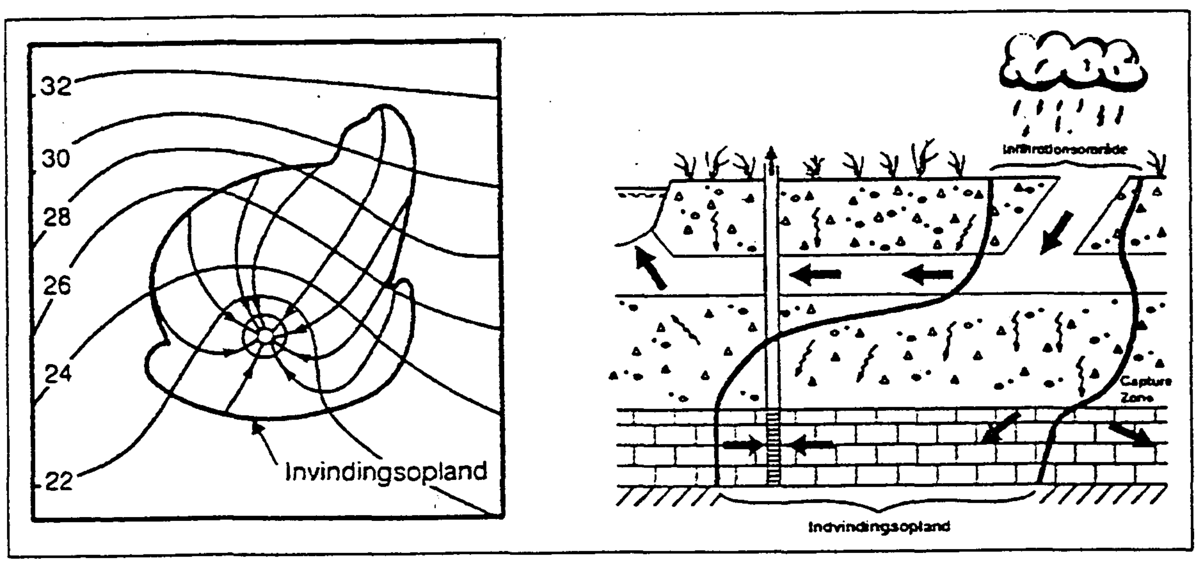
skellig side utroligt meget på, men området er jo egentligt kun interessant i forbindelse med en forureningssituation, hvor forureningen er nået helt ned i det aktuelle grundvandsmagasin.

Infiltrationsområdet for en given boring eller gruppe af boringer kan defineres som det område, hvor det oppumpede grundvand dannes via infiltration gennem jordens umættede zone eller via infiltration fra vådområder og vandløb, se Figur 1b. Dette område kan sagtens - som det vil fremgå af det følgende - ligge langt fra boringerne og helt uden for indvindingsoplandet. Det er dette område, der er interessant i forbindelse med beskyttelsen af grundvandet, idet drikkevandskvaliteten direkte kan påvirkes af kvaliteten af det infiltrerende vand.

Det engelske begreb capture zone er i virkeligheden en kombination af indvindingsopland og infiltrationsområde, idet capture zone kan defineres som det (tre-dimensionale) område indenfor hvilket vandet strømmer til en given boring eller kildeplads. Her kan der altså være tale om et område, der strækker sig over flere geologiske lag, og specielt i forbindelse med forureningsundersøgelser er dette område interessant, se Figur 1b.




Figur 1 Definition af indvindingsopland, infiltrationsoråde og »capture zone«.

# Metoder til bestemmelse af grundvandets sårbarhed

Grundvandets sårbarhed er en svær størrelse at måle og definere. Dels skal der tages stilling til sårbarhed overfor hvad, dels skal der tages stilling til sårbarhed - tid og sted. Det har været en tradition i Danmark, at grundvandets sårbarhed relateres til tykkelsen af det over grundvandsmagasinet (eventuelt) beliggende dæklag typisk bestående af moræneler. Dette er alt andet lige også et relativt mål for, hvor godt beskyttet et grundvandsmagasin er, idet en større tykkelse alt andet lige beskytter bedre end en mindre. Dette »alt andet lige« dækker dog over så mange forskellige forhold, at det i mange tilfælde virker uforsvarligt at anvende dæklagstykkelser alene i forbindelse med vurderingen af sårbarhed. Forhold som ligeledes burde indgå i vurderingen af sårbarhed er (listen er formentlig noget længere, men dette er de umiddelbare for forfatteren til dette indlæg):

| | |
|---|---|
| *ler-type:* | moræneler, smeltevandsler osv. og en vurdering af deres hydrauliske egenskaber; |
| *sprækker:* | det er gennem adskillige forskningsprogrammer påvist, at der kan forekomme sprækker til større dybder (10-15 m), som kan transportere relativt store mængder vand og »stof«; |
| *gradientforhold:* | de drivende kræfter i forbindelse med grundvandsdannelse er gradientforholdene i lodret og vandret retning. Deres variation i tid og sted er af stor betydning for vurdering af grundvandets sårbarhed; |
| *transporttider:* | den tid, som er til rådighed til at etablere afværgeforanstaltninger og til rådighed for de forskellige forurenende stoffer til eventuelt at nedbrydes af naturlig vej; |
| *forureningskildens placering og -type:* | lokaliseringen af potentielle forureningskilder har pågået i en årrække og bør ligeledes indgå; |
| *geokemisk forhold:* | forskellige geologiske forekomsters geokemiske forhold sammenholdt med de forurenende stoffers omsætningspotentiale og nedbrydningsforhold. |



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Det er klart, at samtlige disse forhold ikke kan indgå i den første vurdering af grundvandets sårbarhed i et givent område, men at man på sigt må forsøge at indarbejde så mange af disse forhold som muligt. Det er ligeledes klart, at en grundig vurdering af sårbarheden i langt de fleste tilfælde ikke kan foretages ud fra eksisterende informationer, og at det derfor er nødvendigt at tilrettelægge og gennemføre et målrettet dataindsamlingsprogram.

Anvendelsen af EDB-modellering i forbindelse med vurderingen af grundvandets sårbarhed kan dække over forskellige mere eller mindre komplekse metoder, men det følgende vil udelukkende handle om anvendelsen af grundvandsmodeller eller hydrologiske modeller. En grundvandsmodel er et værktøj, som kan samle og bearbejde alle tilgængelige informationer på konsistent vis inklusive tykkelser af dæklag, deres hydrauliske egenskaber, sprækker, gradientforholdene og deres variation i tid og sted osv. Derfor kan modelanvendelsen under alle omstændigheder give en bedre vurdering af grundvandets sårbarhed end en vurdering, som kun indeholder »dæklagstykkelsen«. Med anvendelsen af en grundvandsmodel får man i mange tilfælde enten af- eller bekræftet forskellige teorier omkring de nævnte forhold, og man kan få kvantificeret deres indbyrdes vigtighed.

## Modellering af grundvandsdannelsen

Grundvandsdannelsen og dens variation i tid og sted er en essentiel størrelse i forbindelse med vurderingen af grundvandets sårbarhed. Dette forhold er styrende for valget af grundvandsmodel, idet en fejlvurdering af grundvandsdannelsen vil få konsekvenser for de senere beregninger.

Som tidligere beskrevet kan en grundvandsmodel udnytte langt de fleste tilgængelige informationer på en konsistent og integreret måde. Grundvandsmodelleringen begynder med opstilling af en såkaldt *konceptuel model for det aktuelle område,* hvori bl.a. indgår en vurdering af de geologiske og hydrogeologiske forhold med henblik på at fastlægge



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områdets udstrækning, overordnede strømningsforhold, randbetingelser til modellen osv. Den konceptuelle model bør endvidere indeholde forslag til kalibreringsmetode og -nøjagtighed.

Efterfølgende skal der opstilles en *geologisk og hydrogeologisk model* for det pågældende område, hvori bl.a. indgår fortolkning af boringsoplysninger, geofysiske undersøgelser og hydrogeologiske undersøgelser og tests samt anden viden om de hydrauliske forhold i området.

På basis af den geologiske og hydrogeologiske model kan *grundvandsmodellen* konstrueres, idet bl.a. antallet af beregningslag, randbetingelser og numeriske kriterier bestemmes. Der skal i den forbindelse fokuseres specielt på randbetingelserne, idet det ikke har nogen mening at angive f.eks. 20 mm årlig nedsivning over hele området, når man efterfølgende skal vurdere på grundvandsdannelsen. Der må nødvendigvis være en beregning af grundvandsdannelsen enten ud fra en beregning af nedbør / fordampning / overfladisk afstrømning eller ud fra en beregning på baggrund af den arealfordelte netto-nedbør. En metode, hvor jordbundsforhold, arealanvendelse og meteorologiske målinger indgår direkte, vil være at foretrække, som det f.eks. foregår i den af DHI udviklede model - MIKE SHE (Storm og Refsgaard, 1996).

Grundvandsmodellen skal nødvendigvis *kalibreres*, idet f.eks. beregnede potentialer sammenstilles med målte ditto, og de hydrauliske parametre justeres indtil tilfredsstillende overensstemmelse er opnået. Det kan i denne fase være nødvendigt at gå tilbage til den konceptuelle model eller den geologiske model og justere på opfattelsen af området, for at opnå overensstemmelse mellem beregninger og målinger.

Modellen bør endvidere *valideres* f.eks. mod målte data fra en anden periode for at vurdere modellens egnethed til at forudsige forskellige hændelser.

I integrerede hydrologiske modeller foretages beregninger af vandstrømninger og opmagasinering for hele landfasen af det hydrologiske kredsløb. Med disse modeller er det muligt at kalibrere og validere modellerne mod observationer af vandløbsafstrømning og grundvands-



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stande, samt desuden mod jordfugtighed, drænvandsafstrømning mv., hvis sådanne eksisterer for det pågældende opland.

Usikkerheden ved bestemmelsen af grundvandsdannelsen ved hjælp af en model afhænger dels af informationstætheden i forhold til variabiliteten i området, dels af påliděligheden af de anvendte informationer. Den kan dog blive væsentlig mindre end samtlige andre metoder, idet flere uafhængige observerede data og deres dynamiske variation kan indgå i kalibreringen og valideringen modellen.

## Modellering af sårbarhed

Med en velkalibreret model kan sårbarheden for et grundvandsmagasin kvantificeres og de områder, som bidrager mest til grundvandsdannelsen kan udpeges. Én metode til dette er alene at se på, i hvilke områder der foregår nedsivning fra de øverste jordlag. Imidlertid giver dette - som det vil fremgå af det følgende - ikke mulighed for at prioritere de enkelte områder i forhold til hinanden, og man risikerer at visse områder ikke »kommer med«. Under mange forhold er der nemlig stor variation både i tid og sted af infiltration, ligesom en del af det infiltrerende vand i mange tilfælde ikke når ned til det grundvandsmagasin, som anvendes til grundvandsindvinding, men i stedet strømmer af til recipienter eller tilsvarende fra øvre magasiner. Figur 2 viser f.eks. beregnede nedsivnings- og udsivningsområder i en sommer- og en vintersituation fra et »moræneopland« (Kristiansstad, Sverige).

Vælger man i stedet at anvende en partikeltransportmodel til belysning af infiltrationsområderne, får man et redskab, der dels kan integrere de stedlige og tidslige variationer og dels kan bidrage med at prioritere infiltrationsområderne. Desuden kan en sådan model anvendes til at vurdere transporttider til indvindingsboringer og i grundvandssystemet i det hele taget. I en partikeltransportmodel slippes et antal partikler løs, deres fødselstid og -sted registreres, og man følger deres position under beregningen.

Deres vej er bestemt af en beregning med en grundvandsstrømningsmodel, som derved afspejler alle kendte forhold i grundvandsmagasi-



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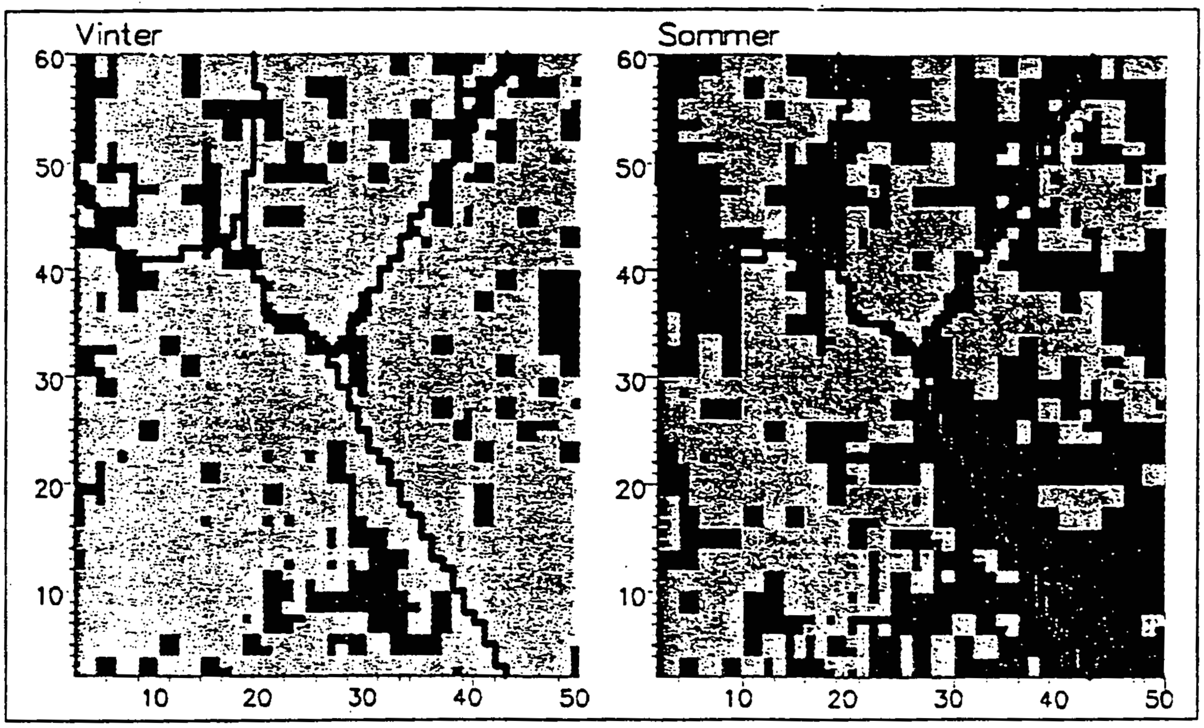


Figur 2: Beregnede infiltrations- og udsivningsområder på forskellige årstider. Kristiansstad, Sverige. De røde og blå felter viser områder med henholdsvis udsivning og infiltration i de øverste jordlag.

net. Ved at registrere, hvor de partikler som ender i indvindingsboringerne er født, kan man få et billede af, hvor grundvandet til de enkelte indvindingsboringer eller -kildepladser er dannet, og hvilken transportbane de har foretaget. Man kan ligeledes vurdere, hvor gammelt det oppumpede vand er, ligesom man kan vurdere transporttider til den enkelte boring. Det sidste er interessant i tilfælde, hvor der er konstateret en forurening af en boring, og man vil undersøge hvem »synderen« kan være, idet man kan sammenholde kort over mulige syndere med et tidsrelateret billede af infiltrationsområdet.

## Eksempler på anvendelse

I det følgende er der givet 2 eksempler på anvendelse af en grundvandsmodel til bestemmelse af grundvandets sårbarhed på forskellige skala.



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## Regionale indvindingsoplande

Fyns Amt og Odense Vandselskab anmodede i sommeren 1996 Dansk Hydraulisk Institut (DHI) om at udpege indvindingsoplandene til de største kildepladser i området i nærheden af Odense i forbindelse med Amtets regionplankortlægning. DHI har gennem tidligere projekter opstillet, kalibreret og anvendt en MIKE SHE baseret hydrologisk model til vurdering af konsekvenser for grundvand og vandløb/vådområder af at etablere en ny kildeplads ved Nr. Søby, Refsgaard (1993), Fyns Amt og Odense Vandselskab (1996). Det var derfor naturligt at tage udgangspunkt i denne model.

Modellen, som dækker det hydrologiske opland til Odense Å samt visse nabooplande - i alt ca. 1000 km$^2$ opløst i et 500 m beregningsnet - indeholder bl.a. en tre-dimensional beskrivelse af grundvandsstrømningerne. I den matematiske model er geologien simplificeret til en 5-lags model med et øvre og et nedre grundvandsmagasin adskilt af aquitarder og med et tyndt lag øverst, som indeholder den umættede zone. Modellen er kalibreret og valideret mod tidsserier af grundvandspotentialer og vandløbsafstrømninger for en 20-årig periode indeholdende både meget tørre og meget våde perioder. Den forventes som sådan at kunne give et meget realistisk billede af de faktisk forekommende strømningsforhold. Modellen indeholder ligeledes en egentlig beregning af »nettonedbøren« og de tids- og arealmæssigt varierende infiltrationsforhold.

Der er udpeget 15 kildepladser, hvortil indvindingsoplandene ønskes beregnet. Fra alle kildepladser foregår der en årlig indvinding på mere end 200.000 m$^3$. Beregningen af oplandene foregår med det nyudviklede partikeltransportmodul til MIKE SHE, idet der først udvælges en repræsentativ periode, hvorfra strømningsberegningerne skal danne basis for de efterfølgende transportberegninger. I dette tilfælde er året 1994 valgt som et repræsentativt år, og transportberegningerne anvender gentagne gange årstidsvariationen i strømningsforhold. Herefter tildeles beregningselementerne et antal partikler, og under simuleringen bevæger disse sig med partikelhastigheden svarende til de aktuelle flux og porøsitetsforhold. Programmet holder styr på, hvilke partikler der havner i én af de udvalgte indvindingsboringer, dvs. hvor kommer den-



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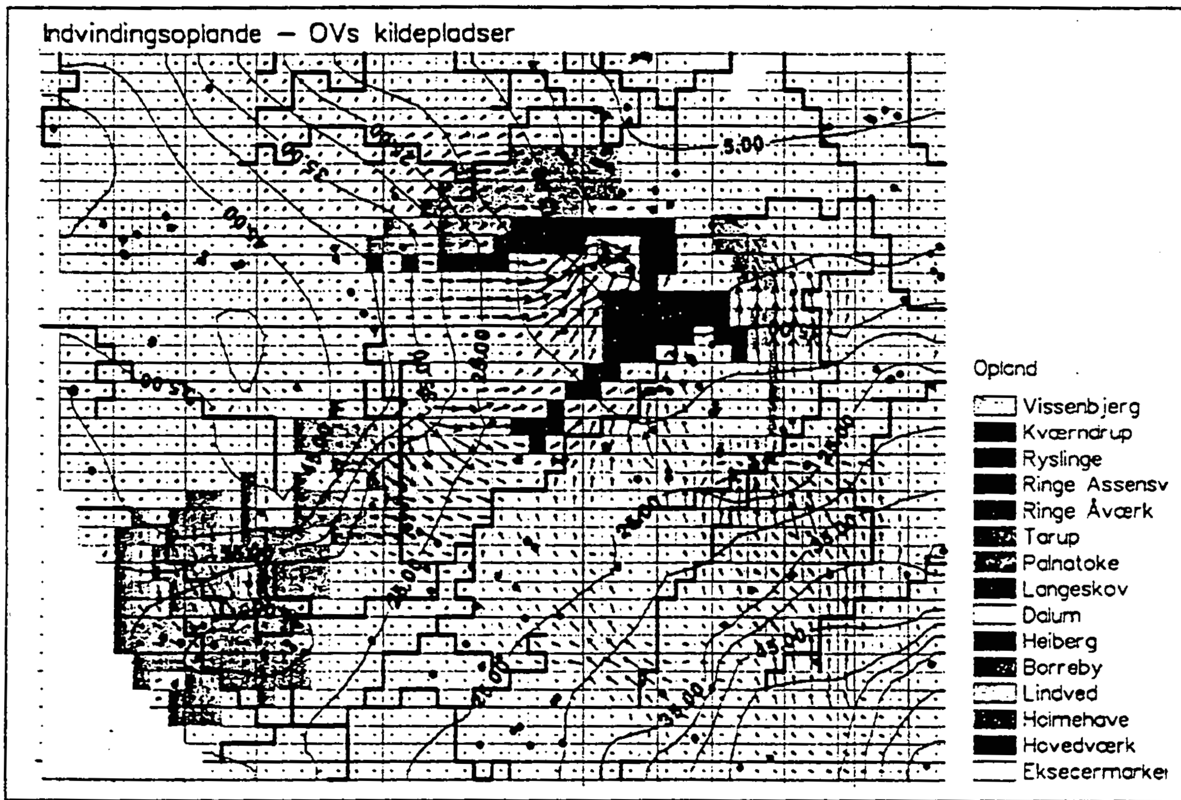
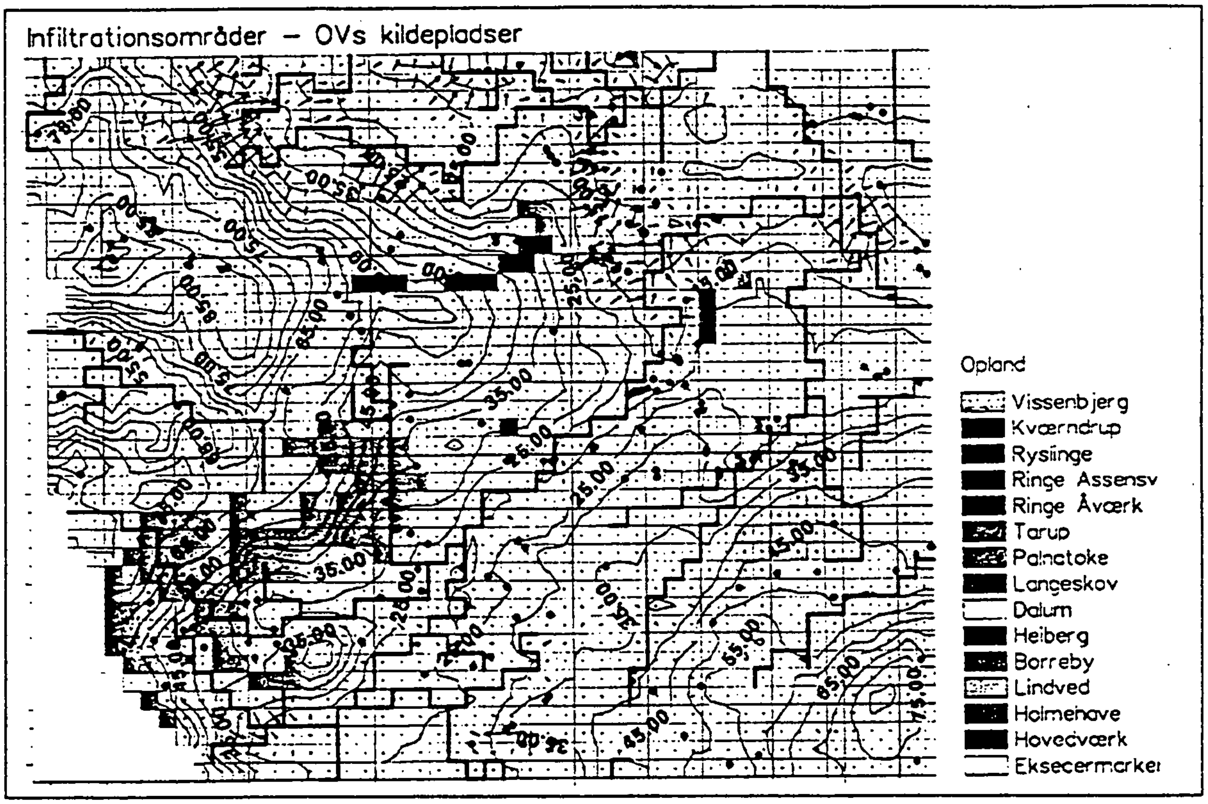
ne partikel fra og hvornår bliver den pumpet op. Resultatet af beregningen er således et tre-dimensionalt, tidsrelateret indvindingsopland (capture zone).

Konkret kan der ud fra beregningerne produceres figurer, der viser dels indvindingsoplandene til de enkelte kildepladser, se Figur 3, dels infiltrationsområderne, til de samme kildepladser, se Figur 4. Beregningerne er foretaget i en 75 års periode, hvilket svarer til, at der er opnået semistationære forhold med hensyn til capture zones.




Figur 3: Beregnede indvindingsoplande (i det nedre grundvandsmagasin) til de største kildepladser til Odense Vandforsyning. Det beregnede potentialebillede og strømningsretninger og -hastigheder er ligeledes vist.

Som det fremgår af Figur 3, er udstrækningen af indvindingsoplandene noget mere kompleks, end man umiddelbart ville forestille sig ved anvendelse af mere simple beregningsprincipper. I flere af oplandene er der »huller«, hvilket skyldes, at lokale indvindinger og/eller lokale geologiske og hydrogeologiske forhold betinger, at vandet fra disse områder ikke strømmer til de udpegede indvindingsboringer. Oplandet til »Hovedværket« strækker sig uden om oplandet til »Eksercermarken« i en form, som de færreste ville kunne beregne eller optegne ud fra potentiale- og transmissivitetsforhold. I det hele taget vil det formentlig være umuligt at anvende simple metoder til bestemmelse af indvindingsoplande, når strømningsforholdene er styret af en kompliceret indbyrdes »kamp« om vandet mellem de forskellige kildepladser.




Figur 4: Beregnede infiltrationsområder, som bidrager mest til grundvandsdannelsen til Odense Vandselskabs indvindingsboringer. Det beregnede potentialebillede og strømningsretninger og hastigheder i de øverste jordlag er ligeledes vist.

Figur 4 viser de vigtigste infiltrationsområder, som her er defineret som de områder i det øverste beregningslag, hvor mindst én ud af fire partikler finder ned til indvindingsboringerne. Med et andet antal partikler kan der bestemmes et større eller mindre areal, idet f.eks. 10 partikler giver anledning til et større areal, mens f.eks. 2 partikler giver anledning





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til et mindre areal, svarende til at man registrerer arealer, hvorfra henholdsvis mindst ca. 10 og mindst ca. 50% af vandet infiltrerer til indvindingsboringerne. Der kan på denne måde foretages en prioritering af beskyttelsen af infiltrationsområderne.

Det fremgår heraf, at disse områder er endnu mere komplekse og har en langt mindre udstrækning, og i flere tilfælde ikke engang ligger inden for indvindingsoplandene. Det komplekse billede af infiltrationsområderne opstår bl.a., fordi der foregår strømninger i de øvre dele af grundvandszonen til vandløb og vådområder og til eventuelle lokale indvindingsboringer, som er filtersat i det øvre grundvandsmagasin.

## Lokale indvindingsoplande

Birkerød Vandværk I/S anmodede i foråret 1996 ligeledes DHI om at vurdere en truende grundvandsforurening med henblik på at optimere afværgetiltagene og forudsige de fremtidige konsekvenser for Vandværket. Opgaven indebar bl.a. opstillingen af en grundvandsmodel for kildepladsen, som består af 10 boringer med en samlet indvinding på ca. 1.6 mill. $m^3$/år, Birkerød Vandværk (1996). Modellen, som dækker det formodede indvindingsopland - i alt ca. 12 $km^2$ opløst i et 50 m beregningsnet - er begrænset til at indeholde grundvandszonen og visse styrende overfladerecipienter. De kvartære aflejringer udgøres øverst af et 5 til 20 m tykt lag af moræneler, som underlejres af et regionalt sand-, grus- og siltlag med en tykkelse på op til 40 m. Disse aflejringer underlejres i visse dele af området af et tyndt lag af smeltevandsler, som har ringe vandgennemtrængelighed, mens det i andre områder direkte underlejres af præ-kvartære kalkaflejringer, som udgør det primære reservoir for vandindvinding.

Modellen er i vertikalen opdelt i 10 beregningslag, hvilket giver en realistisk beregning af de tre-dimensionale strømningsforhold, der i høj grad er styret af indvindingsstrukturen, de omkringliggende recipienter (Sjælsø) og udbredelsen af det tynde lag af smeltevandsler. Der er anvendt samme fremgangsmåde som i Odense-projektet til bestemmelse af indvindingsoplande og infiltrationsområder.

Med den noget finere diskretisering af området i 50 m beregningselementer har det været muligt at bestemme indvindingsoplandene og



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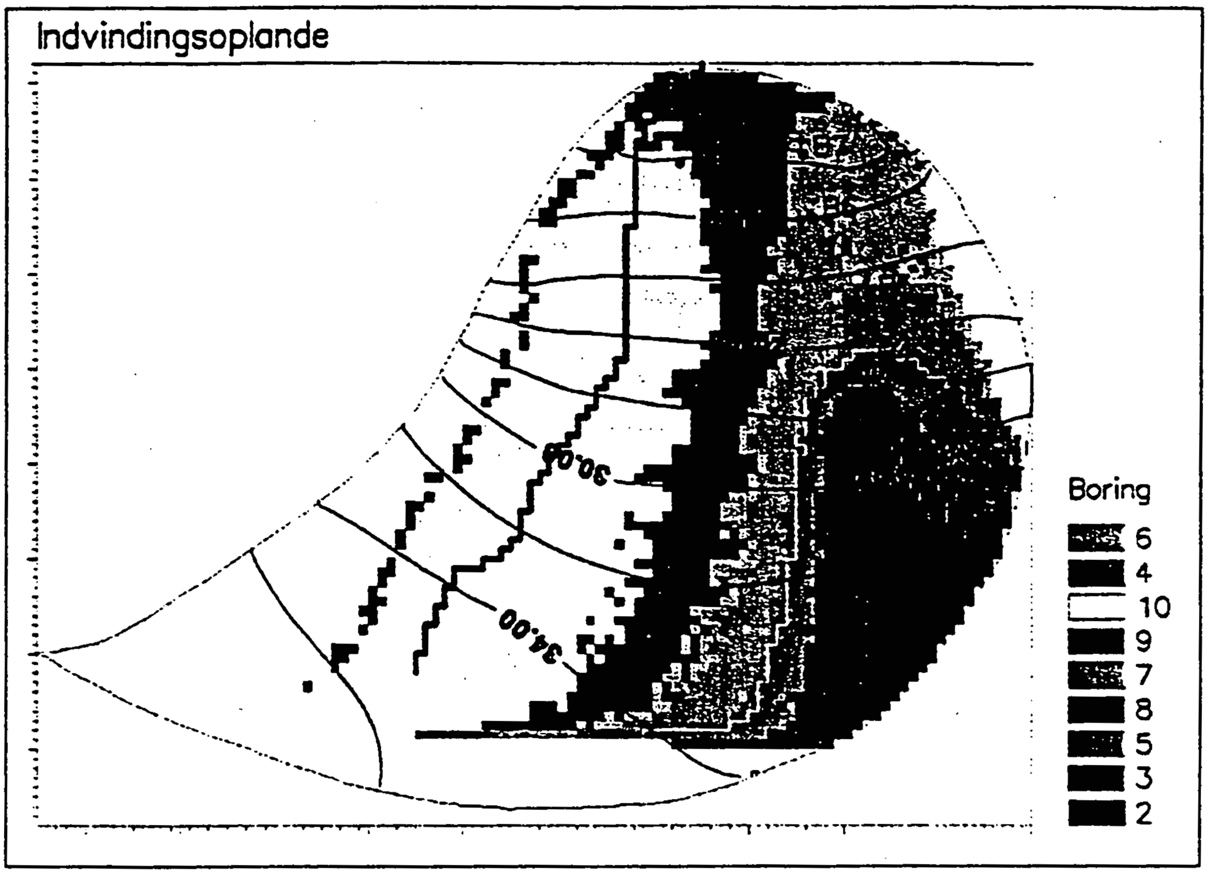


Figur 5: Beregnede indvindingsoplande (i det nedre grundvandsmagasin) til de enkelte boringer til Birkerød Vandværk.

infiltrationsområderne til de enkelte boringer på kildepladsen, som vist i Figur 5 og 6. Indvindingsoplandene viser et meget komplekst billede af, hvilke områder der bidrager med vand til de enkelte boringer - se f.eks. indvindingsoplandene til B4, B5 og B9, som ud over at være »klemt« mellem andre oplande strækker sig uden om disse i nogle meget komplekse mønstre.

Infiltrationsområderne er ikke mindre komplekse, og er som det fremgår af Figur 6, beliggende langt fra selve indvindingsboringerne. Dette skyldes, at det vand, der infiltrerer på selve kildepladsen, strømmer via højtliggende sandlag direkte mod Sjælsø og dermed ikke bidrager til grundvandsdannelsen til de nedre magasiner på kildepladsen. Hvis man vil beskytte vandkvaliteten i indvindingsboringerne, skal man altså sikre, at det i infiltrationsområderne dannede grundvand har en god





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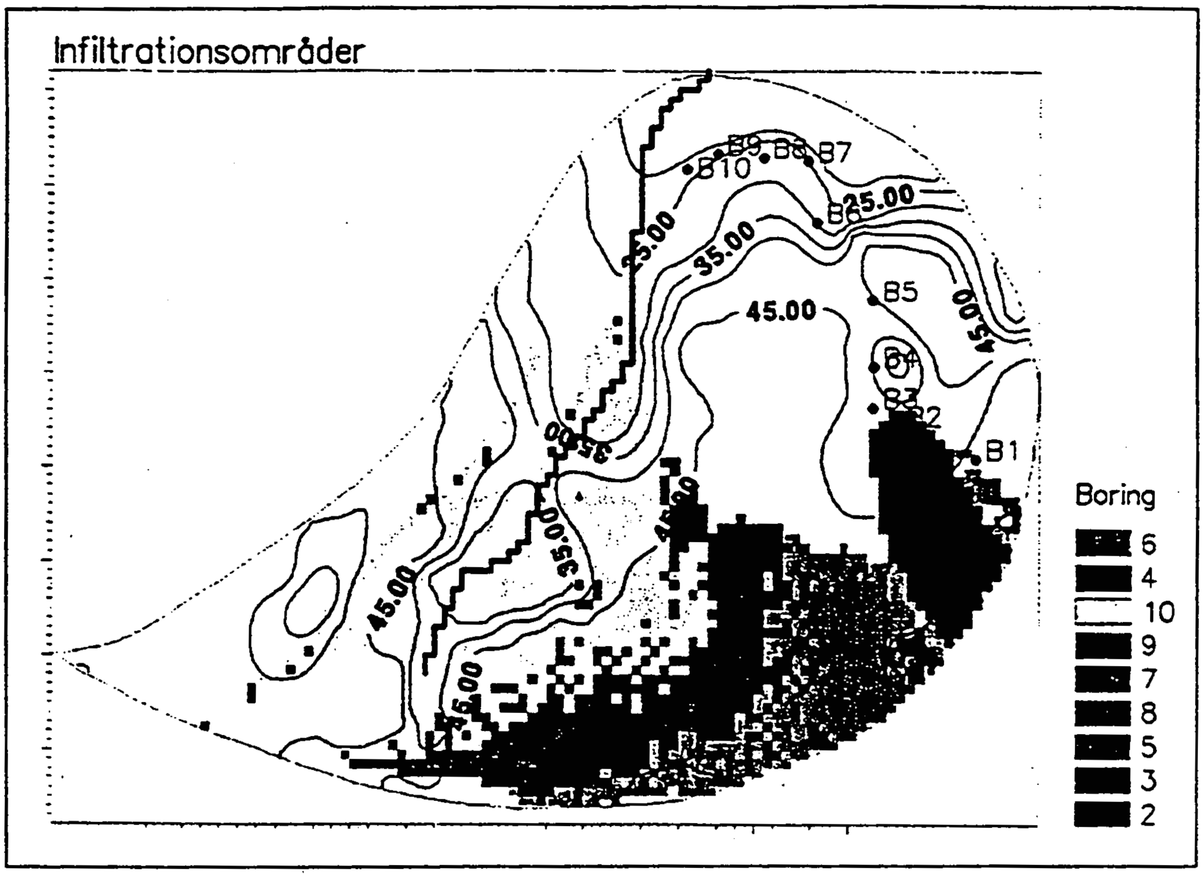


Figur 6: Beregnede infiltrationsområder, som bidrager mest til grundvandsdannelsen til Birkerød Vandværks indvindingsboringer.

vandkvalitet. Det er altså i dette tilfælde lidt svært at se begrundelsen for at fokusere på den såkaldte kildepladszone, som er defineret som området mindre end 500 m fra indvindingsboringerne.

En anden og måske lige så vigtig anvendelse af en partikel-model er til at bestemme transporttider til eller fra et givet område eller boring. Nedenstående Figur 7 viser et eksempel på, hvorledes modellen kan bidrage til at bestemme »forurenings-synderen« overfor forurening i en boring - i dette tilfælde Birkerøds Vandværks boring 2. Ved at beregne transporttiden fra jordoverfladen til boringen og sammenholde med mulige forureningskilder, kan »synderen« udpeges med rimelig stor sandsynlighed og en indsats mod yderligere forurening eller afværgeforanstaltninger kan iværksættes.





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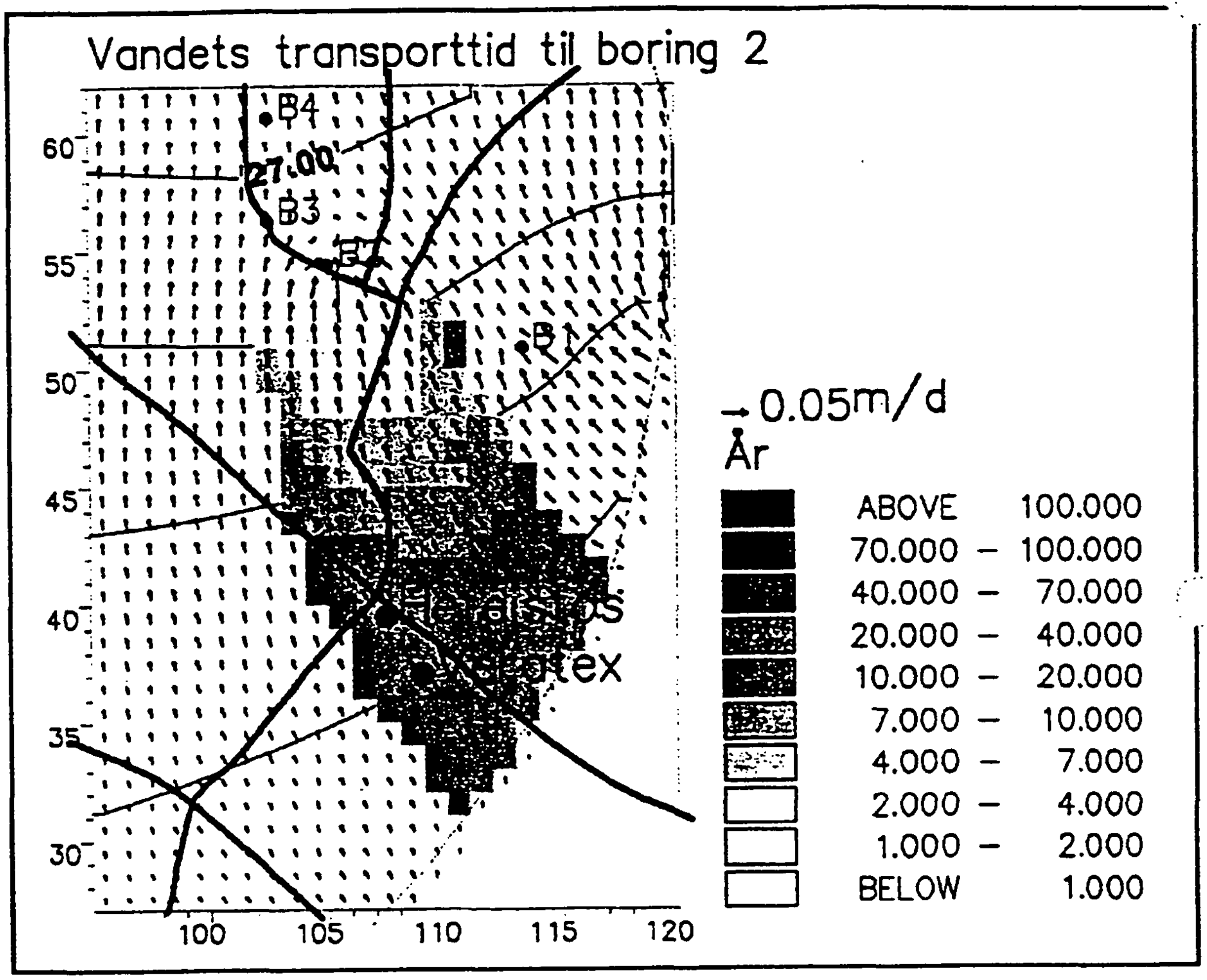


Figur 7: Beregnede transporttider fra jordoverfladen til Birkerød Vandværks Boring 2.

# Konklusioner

Ovenstående eksempler viser anvendeligheden af en numerisk partikel-transportmodel kombineret med en strømningsmodel i forbindelse med udpegningen af indvindingsoplande og infiltrationsområder. Det er formentlig det eneste redskab, der kan bestemme de komplekse områder under hensyntagen til aktuelle komplicerede forhold omkring geologi, hydrogeologi, indvindingsmønster osv. Selv om resultaterne - ligesom andre metoder - skal tages med de forbehold, som afspejles af usikkerhederne på bestemmelsen af de strømnings- og transportmæssige parametre i en numerisk model, kan dette være et vigtigt redskab i forbindelse med prioriteringen af indsatsen for en bedre grundvandskvalitet i fremtiden.

Partikelmodellen kan endvidere anvendes til at bestemme tidsrelaterede oplande til enkeltboringer, hvilket er en stor hjælp i forbindelse med udpegningen af forureningssyndere, idet disse informationer sammenholdt med informationer omkring mulige forureningskilder og perioder, hvor disse har kunnet forurene grundvandet, kan udpege forureningssyndere med stor sandsynlighed.

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# Europe goes data divining

It's difficult to collect, not always very clean and some parts of Europe have much more of it than others. STEFAN KLEESCHULTE and JENS CHRISTIAN REFSGAARD set off in search of good quality hydrological data on a pan-European scale.

Hydrological modelling depends on a whole range of inputs to create effective and accurate simulations. Both the spatial and non-spatial data which feeds into the models must be sufficiently detailed to provide useful outputs. But that has to be balanced against the need for manageable quantities of data and reasonable processing timescales.

Water does not respect national boundaries. However, it is still difficult to find relevant datasets covering the whole of Europe in a complete and consistent way. Apart from European Commission services like EUROSTAT and the European Environment Agency, few organisations in Europe provide environmental datasets that cross national borders. Even where users can find suitable data, it is often unusable because it is protected by copyright.

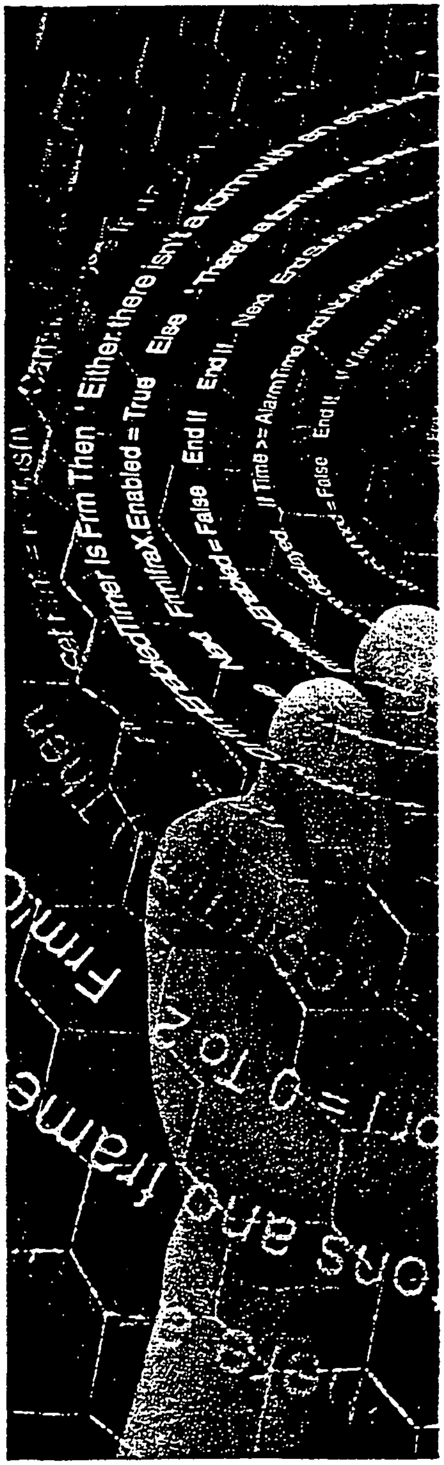
If hydrological models are to be applied at a continental level, they have to be able to cope with multiple data sources, and the cumulative effects of inconsistency or uncertainty in data. A pan-European consortium of academic and private sector organisations was set up to investigate how disparate data sources affect Spatial Decision Support Systems (SDSS). The initiative was named UncerSDSS.

## Laying the groundwork

The project focused particularly on groundwater contamination from aluminium and nitrates. In Europe, groundwater supplies most of the drinking water for humans and animals. When fertilisers and manure are used on farmland, potentially harmful nitrates leach into the groundwater, while rain and wind carry high levels of acidic substances which dissolve aluminium and carry it into the water.

*Hydrology n. The study of water, including rain, snow and water on the earth's surface, covering its properties, distribution, conservation, utilisation, etc.*

The UncerSDSS project set out to develop a pan-European hydrological modelling application, testing it in Denmark and the Netherlands. The team started by defining a SDSS as a combination of a hydrologic model and a GIS. If the output from the system was to be useful for pan-European applications, the input data needed to meet certain criteria. It had to be available for the whole of Europe and harmonised across regional and national boundaries, to ensure consistent results. The GIS data had to be accessible from a single, seamless and integrated database. And ideally, the entire dataset should be available from a single source.



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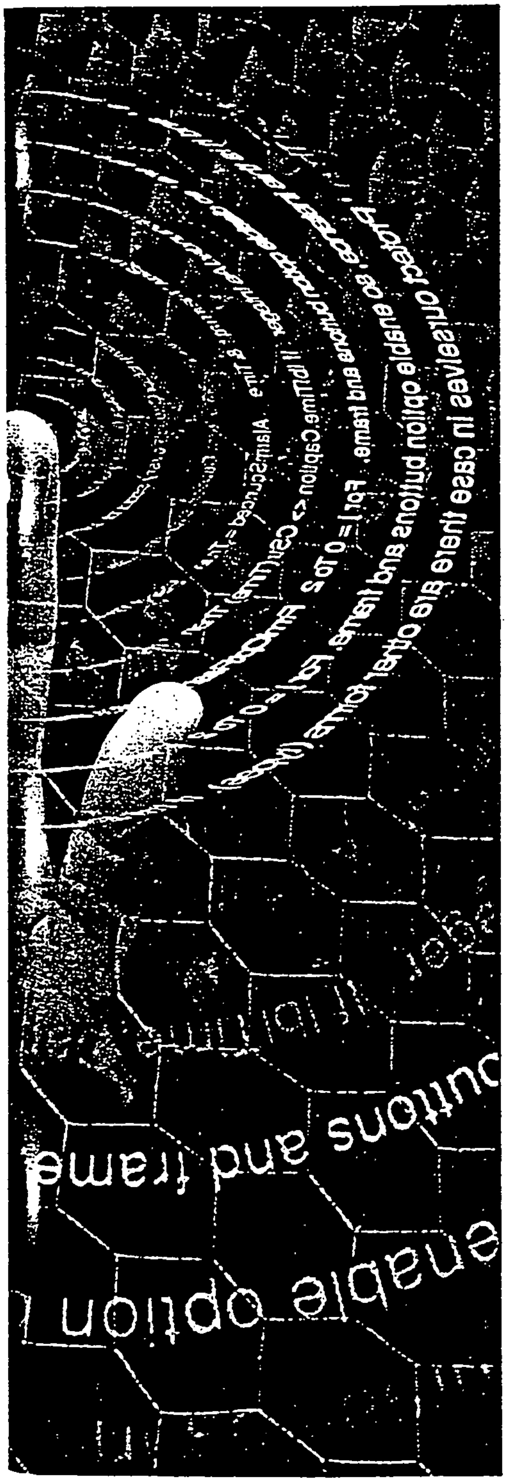
What was meant by a seamless database? In this case, it had to use a common naming system for all the objects in the database. National datasets often use different naming systems for the same object, depending on its original purpose and use. So, for instance, land-cover might be classified differently, depending on whether it was originally being used to support water protection or the conservation of rare species and habitats. Another critical feature of an integrated database is that it uses the same scale and level of detail throughout. And finally, thematic and geometric treatment need to be consistent, to avoid edge-matching problems across boundaries.



Data from national sources are not generally harmonised in this way across national boundaries, and data collection methods are often based on different basic assumptions, so datasets are difficult to compare. This meant that, as far as possible, the project team avoided using national data. On the other hand, the resolution of datasets offering worldwide coverage was usually too coarse to provide enough detail.

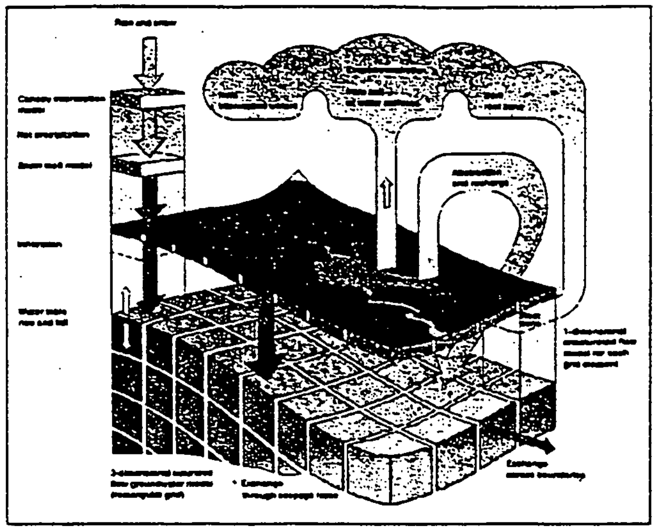


**Figure 1: The Mike SHE model. A distributed, physically-based hydrological modelling system applicable to a wide range of water resources problems related to managing surface and ground water.**

With these factors in mind, the UncerSDSS team set out to find what good-quality data really was available at a European level. When the project started in early 1995, online search tools and data catalogues were only just starting to emerge, so the team concentrated initially on traditional data sources, particularly analogue and digital data catalogues from international map publishers and European institutions.

The team also approached public and private organisations active on the European data market via the Internet, and searched their on-line data lists for useful information. The Internet was also systematically trawled using general search tools (such as AltaVista) with a large number of relevant keywords. This led to a large selection of specialised sites, whose information contents were assessed in more detail at the next stage.

Searching all these potential data sources was very time-consuming. It proved easy to find good-quality datasets on land cover and soil types, as well as many statistics. But the data for other modelling requirements—such as soil hydraulic properties and organic matter—proved more elusive. These databases were very often compiled as part of a specific project, so they were not necessarily available publicly. Locating them was more a matter of luck than systematic screening.

### Table 1: project members

- Danish Hydraulic Institute, Denmark
- Royal Veterinary and Agricultural University, Denmark
- DLO Winand Staring Centre, The Netherlands
- University of Amsterdam, The Netherlands
- Chalmers University of Technology, Sweden
- Techna Consult, Belgium
- Geographic Information Management SA, Luxembourg

UncerSDSS was partly funded by the EC Environment and Climate Research Programme.

## Quality in, quality out

Hydrological models need a wide range of data inputs. For the UncerSDSS project, two models were used (see figures 1 and 2). Mike SHE (Système Hydrologique Européenne) is a distributed, physically based hydrological modelling system related to surface and ground water management, contamination and soil erosion. The Simulation Model for Acidification's Regional Trends (SMART2) is a simple, one-compartment soil acidification and nutrient cycling model developed to evaluate the effectiveness of emission control strategies.

Neither model was especially developed for the project, but both were developed for use with European data. Each has slightly different data needs: table 2 summarises their requirements. The second column shows potential European data sources which could have satisfied the need for data, but which—in some cases—did not provide enough detail for one or other of the models, or was not accessible for some reason. The third column shows the actual data source which the project used.

## Table 2: required input data and data sources

| Data | Potential data source identified in European database | Source actually used for modelling |
|---|---|---|
| Atmospheric deposition | EMEP[1] and IIASA[2] information on atmospheric deposition (not yet published) | National data |
| Climatic variables for Denmark | MARS[3] project | National data |
| Climatic variables for The Netherlands | Eurostat GISCO[4] | Eurostat GISCO |
| Crop type | Eurostat – Regional Statistics | Agricultural statistics (1995) for Denmark |
| Fertiliser consumption | Eurostat – Environmental Statistics | Agricultural statistics (1995) for Denmark |
| Geology | Report on groundwater resources (EC, 1982) RIVM[5] – digital map data of report | Report on groundwater resources (EC, 1982) – different country reports |
| Groundwater abstraction | Report on groundwater resources (EC, 1982) RIVM[5] – digital map data of report | Report on groundwater resources (EC, 1982) – different country reports |
| Livestock density | Eurostat – Regional Statistics<br>Eurostat – Eurofarm | Agricultural statistics (1995) for Denmark<br>Agricultural statistics (1995) for Denmark |
| Management practices | SC-DLO[6] report | *Plantedirektoratet* (1996) for Denmark |
| Manure production | Eurostat – Environmental Statistics | Agricultural statistics (1995) for Denmark |
| River cross-sections | Provided by an application developed as part of the project | Provided by an application developed as part of the project |
| River network | DCW[7] | Provided by an application developed as part of the project |
| River run-off | GRDC[8] data | National data |
| Soil organic matter | RIVM[5] report, data for natural soils only | Literature values for Danish arable soils and transfer functions |
| Soil type | GISCO[4] soil map | GISCO[4] soil map |
| Topography | USGS[9]/GISCO[4] | USGS[9]/GISCO[4] |
| Vegetation | EEA[10] – CORINE[11] land cover | EEA[10] – CORINE[11] land cover |

### Abbreviations

1 EMEP – Co-operative Programme for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe
2 IIASA – International Institute For Applied Systems Analysis
3 MARS – Monitoring Agriculture by Remote Sensing
4 GISCO – Geographic Information System of the European Commission
5 RIVM – National Institute of Public Health and the Environment of The Netherlands
6 SC-DLO – Winand Staring Centre, The Netherlands
7 DCW – Digital Chart of the World
8 GRDC – The Global Run-off Data Centre, part of the World Meteorological Office – a database of daily river discharges and station history
9 USGS – United States Geological Survey
10 EEA – European Environment Agency
11 CORINE – Coordination of Information on the Environment

In some instances, the initial source identified by the project was a EUROSTAT dataset. Unfortunately, EUROSTAT's pan-European statistics do not always implement every "NUTS" level for all EU member states. NUTS levels represent the different levels of administrative subdivision for each country: from NUTS 0 (the whole country) to NUTS 5 (individual municipalities). This was a particular problem for the UncerSDSS project, because no statistical data below national level was available for Denmark. The solution was to use the original national statistical dataset which fed the higher-level EUROSTAT database.

Finding appropriate datasets for geology and soil properties proved more difficult. The simulation models demanded very specific parameters, such as annual groundwater withdrawal or recharge, and hydraulic conductivity and organic matter content for agricultural soils. The team found some of this data in European Union (EU) reports and publications, but often the scope of these projects was limited, so the data was not directly applicable at a European scale. In these cases, parameters were modelled on the basis of available data.



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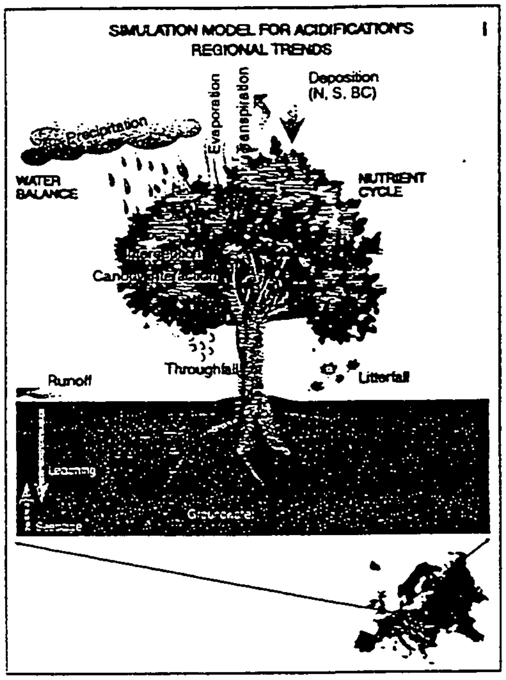


**Figure 2: The SMART2 model concentrates on soil acidification and nutrient cycling.**

Hydraulic conductivity, for instance, was extrapolated from data on soil texture.

When it came to river run-off data, this is normally easily available from the Global Run-off Data Centre (GRDC) which holds daily or monthly discharge values for most of the world's main rivers. However, this database did not hold data on the relatively small rivers in the Danish test area, so national sources had to be used instead.

Climatic data is available through national meteorological offices, and the measurements are comparable all over Europe, although quotations for providing appropriate data for the UncerSDSS project varied widely from country to country. At EU level, EUROSTAT offers a dataset with climatic variables for some 5,000 stations. Unfortunately, the time resolution for this data was too coarse, so it could not be used, and copyright prevented the team using a dataset with better resolution from the Monitoring Agriculture by Remote Sensing (MARS) database.

## The learning curve

The biggest obstacle to using European databases is generally copyright. EU organisations often collect information from member states for a specific purpose, and the resulting database is not available to other users, even within the Commission. Experience from the project also shows that many so-called "European" databases are in fact only a collection of disparate national datasets held in a common data repository. This kind of data is not always usable in cross-border projects.

EU statistics depend on the NUTS levels used for the different member states. In small countries, data is often already aggregated at a national level (NUTS 0), while in larger states there are often sub-divisions. More detailed statistical information is generally available from national statistical offices. However, it is sensible only to use national datasets where the data quality is reliable, and standardised data collection methods have been used.

In a project like this, the ideal solution is a European clearing house providing access to digital spatial data through metadata. Operating as a detailed catalogue service, with links to spatial data providers and browsers, a clearing house would enable users directly to download appropriate digital datasets in different formats.

In the meantime, though, the UncerSDSS project shows that it is possible to build hydrological models using pan-European datasets. However, there is still a long way to go in providing appropriate and easily accessible data, despite the continuing efforts of EU institutions like EUROSTAT. G

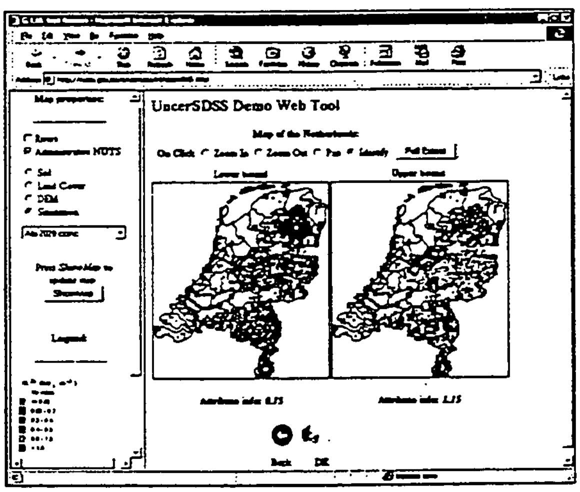


**Figure 3: UncerSDSS on the Net. The online viewing tool displays input data and simulation results, using a GIS tool built with ESRI's MapObjects. The site can be found at http://projects.gim.lu/uncersdss.**

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# SHYLOC: System for hydrology using land observation for model calibration

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ABSTRACT: Only with deterministic hydrological models is it possible to determine the environmental impact of potential changes to protected wetlands and to understand their functions in order to manage them properly. So far the use of these models has been slow due to limitations in the models themselves, the large amount of manpower required to set them up and the limited amount of data to calibrate them.

The SHYLOC project aims at resolving all these problems through improvement of the models themselves and the use of an innovative method of using satellite images to determine surface water storage in networks of surface water channels.

## 1 BACKGROUND AND OBJECTIVES

### *1.1 Wetland Loss*

Around the world wetlands are being lost and degraded as economic development results in increasing pressure to drain and re-claim land for agricultural, industrial and other uses.

The causes of wetland loss and degradation are many and varied. Some are deliberate, others the result of decisions taken without a full appreciation of the importance of wetlands. Hollis (1992) divided the causes into: root causes (which may include population pressure, lack of public and political awareness of wetland importance, absence of political will for wetland conservation, over-centralised planning procedures, and financial policies), external factors (such as policies and activities of external organisations and Development Assistance Agencies), immediate causes (including weak conservation institutions, sectoral organisation of decision making; deficient, or in many cases, no application of environmental impact and cost-benefit analysis legislation, lack of trained personnel and limited international pressure) and finally obvious and apparent causes (agriculture, pollution, urbanisation, water resource schemes, fisheries, tourism and recreation, erosion and sedimentation).

"Wetlands as wastelands" has been a powerful metaphor in the past and since wetlands have been believed to have no value they have been destroyed indiscriminately throughout Europe. South-west England has lost 92% of its wet pastures since 1900 (Denny, 1993). During the drought of 1992, over 500 Sites of Special Scientific Interest (SSSI) in England were hydrologically affected by water abstraction. The number of ponds in Hertfordshire, UK has fallen by 50% since 1985 (Hertfordshire County Council, 1987) and the area of grazing marshes in the wider area of the Thames Estuary has fallen by 28,000 ha (65%) since 1930 (Thornton, and Kite, 1990). In France freshwater meadows, bogs and woods once covered 1.3 million ha but this area has been declining by 10,000 ha each year (Baldock, 1989). In Roman Times wetlands covered 10% of Italy. By 1865 this had declined to only 764,000 ha and in 1972 only 190,000 ha remained (Ramsar Bureau, 1990). 60% of the wetlands in the Castille La Mancha region of Spain have been lost with three quarters of this loss (200,000 ha) taking place in the last 25 years (Montes and Bifani, 1989). In Portugal 80% of saltmarshes are threatened by reclamation whilst Greece has already lost 60% of its original wetlands, mainly through the drainage of lakes and marshland for agriculture (Psilovikos, 1988) and 12% of the

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remaining wetlands are threatened by changes in water regime (Zalidis et al., 1997).

### *1.2 The Difficulty of Hydrological Modelling*

Hydrology is the key factor controlling the structure and functioning of any wetland which in turn determines specific ecosystem responses. An understanding of wetland hydrology is therefore a vital prerequisite for successful wetland management.

Hydrologists construct models of river catchments and wetlands in order to understand their functioning so that past events can be explained and future ones predicted. Wetlands present special problems.

- It is often the storage, rather than the flow of water that dominates the water balance.
- Many wetlands in Europe are greatly influenced by rather complicated boundary conditions such as pumping stations, weirs, and sluices. Kaiser et al., 1997 and Al-Khudhairy and Thompson both found that the MIKE-SHE (DHI, 1993) hydrological software was capable of modelling wetlands but that these man-made features needed careful handling.
- The measurements required to calibrate models have often not been made because of their low commercial interest.

Another difficulty with modelling is general to hydrology as a whole: the effort required to collect data, construct a model and to calibrate them is prohibitive. For instance the English Environment Agency rarely construct models for the preparation of their Water Level Management Plans (written statements prepared by an operating authority and endorsed by other interested parties to manage water levels in a prescribed area).

### *1.3 The Use of Remote Sensing in Hydrology*

Bearing this in mind we can now consider the various methods whereby remote sensing can be used to help hydrologists. Books on the subject include that by Engman and Gurney (1991). A number of conferences and workshops have also been organised. Many of the methods discussed show promise for the future but even their proponents admit that the accuracy from sensors based on present satellites is inadequate and require more development before they are usable on a regular basis. Let us consider the various contributions that remote sensing can make in turn.

1) Vegetation and land cover. The effect of changes in land cover on hydrology is well known and generally well accepted. The classification of landcover using remote sensing is widespread. In fact landcover maps for most of Europe, obtained at least in part by remote sensing, are already commercially available. For wetland inventories satellite data is clearly not yet able to classify habitats by life form, water regime, substrate type and water chemistry. Major changes, such as urbanisation or conversion from natural to arable are, of course, visible and satellite remote sensing has already been used to identify such changes in protected areas.
2) Precipitation. Rainfall measurements might be possible with future active microwave satellites but at least 50 years of records from ground-based rain gauges are available across most of Europe so rainfall from remote sensing is not a priority for hydrologists.
3) Runoff. Runoff cannot be measured directly although it can be estimated from land cover and watershed geometry. We have already seen that remote sensing can contribute to landcover. It has been suggested that digital elevation models can, as well, be obtained from stereographic sensors and interferometric techniques (for example Muller et al. 1995) and this will be useful, in particular in areas such as the mouths of estuaries where topography can change rapidly. In the wetlands considered in this paper digital elevation maps obtained by traditional means and available commercially provide an altitude accuracy of about 1.2m.
4) Flood area. The best hope of capturing flood area comes from synethic aperture radar (SAR). They have been used in a research context but their signal is dependent on wind and vegetation as well as the presence of water. A combination of the image with other data is feasible but as yet no operational system exists.
5) Evapotranspiration. This cannot be measured directly by remote sensing and nearly all operational techniques use correlations obtained from ground based meteorological stations to estimate potential evapotranspiration and then combine this with vegetation

characteristics and soil moisture to obtain actual evapotranspiration.

6) Soil moisture. The possibity of measuring soil moisture (ie the integrated quantity of water stored in the root zone) by remote sensing measurements over a large spatial area would be useful for hydrologists. Microwave techniques have a strong potential because of the difference in dielectric constant between wet and dry soil (it changes from about 20 to about 3). But both passive and active systems are hampered by the difficulty in separating the influence of vegetation, surface roughness and soil type from that of moisture. Jackson and Schmugge (1991) note that vegetation water contents in excess of 6 to 7kgm$^{-2}$ can totally obscure the soil surface at the 21cm wavelength.

We have outlined why hydrologists have been both excited and disappointed by the promise of remote sensing. The excitement is due to the twenty year archive of data and the possibility of having a spatial distribution of information. The disappointments are due to the present lack of operational systems using remote sensing to offer more than ground measurements (digital terrain, meteorological data) as well as the poor accuracy (soil moisture) and insufficient frequency (flood mapping) of the remote sensing measurements.

## 2 MIKE-SHE CALCULATIONS

### *2.1 The North Kent marshes*

The foundation work for the SHYLOC project was laid by a number of studies to understand how well standard commercial hydrological codes could model wetlands. The first study was the North Kent marshes. These are located in South East England along the lower Thames estuary and on both sides of the Swale (a tidal channel which separates the Isle of Sheppey from the mainland). These features can be seen clearly in a satellite image (Figure 1). The marshes are largely the product of human activities which go back as early as Roman times. A series of sea defences have been constructed, in the form of embankments and walls, to isolate and protect former salt marshes from the sea. The majority of these are at least several centuries old although some small areas were enclosed as recently as the last 20 or 30 years. During the period 1960-1980 extensive drainage and field amalgamation took place within the marshes to convert them into arable land. Since then little new drainage and conversion has taken place, reflecting the increasing concern of nature conservation organisations about the loss of traditional grazing marshes and the awareness of farmers of the unstable nature of heavy marsh soil after underdrainage.

The drainage system in the North Kent marshes consists of a network of fleets, runnels and ditches inherited from pre-enclosure salt marshes but extensively modified by human activity. These divide the marshes into fields of varying size. Along the sea walls there are a number of tidal flaps (sluices) which let drainage water escape into the sea at low tide and largely exclude salt water inflows at high tide. At present, wetland management focuses on maintaining the traditional water regime of the marshes. The ecological importance of the North Kent Marshes is reflected in their designation as Sites of Special Scientific Interest (SSSI), Ramsar Sites under the Convention on Wetlands of International Importance, and Special Protection Areas (SPA) under the EEC Birds Directive (79/409). The marshes are also within the Ministry of Agriculture, Fisheries and Food's (MAFF) Environmentally Sensitive Areas (ESA) scheme which recognises their importance for birds, nature conservation and archaeological sites.

The hydrology of a large part of the North Kent Marshes on and immediately South of the Isle of Sheppey was summarised by Hollis et al. in 1993 who found the hydrology of the marshes to be rather complicated. The marshes

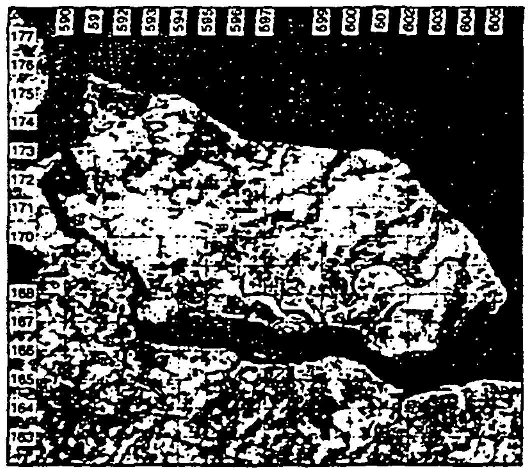


Figure 1 Band 5 of Landsat-TM image, 12 February 1988, showing clearly the Isle of Sheppey and the Swale, image © Eurimage

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could be divided into a number of very small catchments varying in size between 1 and 10km$^2$ and, although there was a limited amount of inflow from uplands in some cases and some outflow through the sluices, the water balance in these catchments was dominated by the rainfall onto the marshes and the evapotranspiration from them. Hollis et al. hypothesised that the ditches were draining the fields in winter and maintaining the water table in summer but considered it difficult to ascertain the exact relationship between the water table in the fields and the level of water in the ditches without further measurements.

In an effort to understand further the hydrology of the marshes Al-Khudhairy et al. (1996) developed a distributed hydrological model of one marshland catchment of about 10km$^2$ using the MIKE-SHE software (DHI, 1993). The catchment they modelled, Bell's creek, was not chosen for its representativeness - in fact the underdraining of the fields made it rather unrepresentative of the most ecologically interesting sites - but because this was the only catchment where calibration measurements could be found. A drainage pump had been operating there for a number of years and records of the pumping rates existed. The underdraining made it difficult to extrapolate the findings to other areas although, of course, the MIKE-SHE model itself did give some indication of its influence. Indeed this was one of the main objectives of a study by Al-Khudhairy and Thompson, 1997, who applied this model f determining the impact of land-use changes on the wetland's functioning. Figure 2 shows the discharge out of the system (using the pumps) firstly for the present situation and secondly as it would be without drainage and thirdly with grassland instead of arable.

They did find MIKE-SHE extremely useful for examining hypotheses and it is difficult to see how this could have been done in a quantitative manner without such a deterministic model. Nevertheless some contrivances had to be used in order to model the various boundary conditions - weirs, tidal sluices - that govern the hydrology of the ditches and hence the marshes themselves.

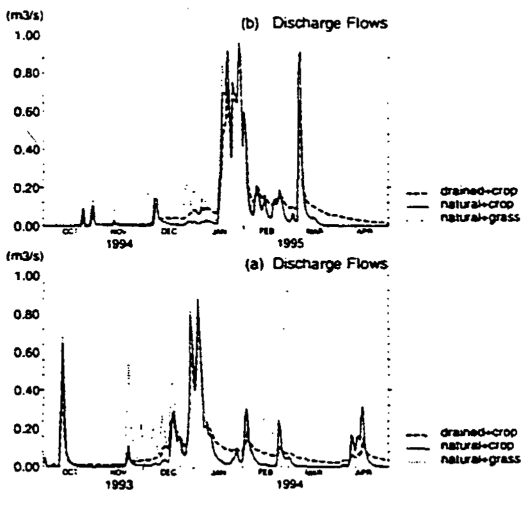


Figure 2 Discharge out of Bell's Creek:

- with present land-use
- assuming these is no underdainage
- without drainage and with grassland instead of barley

*2.2 Lake Karla*

The second test case was Lake Karla in Thessal- Greece. Originally Karla was a closed valley fe.. by runoff from the surrounding hills (figure 3) and flooding from the Pinios river which passes close to the valley entrance. Once in the valley the water could only escape by evaporation and the shallow lake at the valley bottom varied in size depending on the season. During the thirties an embankment was built alongside the Pinios river so floodwater no longer entered the valley. This reduced the catchment area to its present 1000km$^2$ and started the lake's decline. It was reduced in size despite a still considerable runoff from the surrounding hills. As late as the fifties an all-time Greek record of waterfowl was counted on it.

Unfortunately the flooding disturbed agriculture and a study of remedial measures was commissioned. The study suggested draining th lake by drilling a 10km tunnel from the close.. sump end of the valley to the sea at the Pagastikos Gulf near Volos and the construction of a reservoir for flood control and water storage. In

Figure 3 Computer Reconstruction of Lake Karla

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the event only the tunnel was built and this was effectively the end of Lake Karla. But it was not the end of the problems. The agriculture required water and this was obtained from groundwater. The supply could not keep up with demand and the water table has now descended 100 metres and become increasingly saline. Furthermore the drainage channels are polluted with effluent from factories in Larissa and fertilizers from the land. This makes the inhabitants of Volos reluctant to allow discharge into the Pagastikos Gulf where the water is relatively stagnant. Furthermore the area is still prone to flooding. The 8.5 $m^3s^{-1}$ maximum flow from the tunnel is not enough to discharge all floodwater during longer periods of rainfall. So a reservoir is being built to relieve the flooding and to restore pre-existing wetland functions on a longer term.

Various attempts have been made to model the behaviour of Karla. Kaiser et al. (1997) used MIKE-SHE in order to understand groundwater depletion and analyse various restoration options. In this case there was some calibration data - the groundwater levels had been recorded in a number of places and the flow through the tunnel had been monitored. As with North Kent a number of contrivances had to be made in order to model correctly the behaviour of the boundary conditions - especially the tunnel.

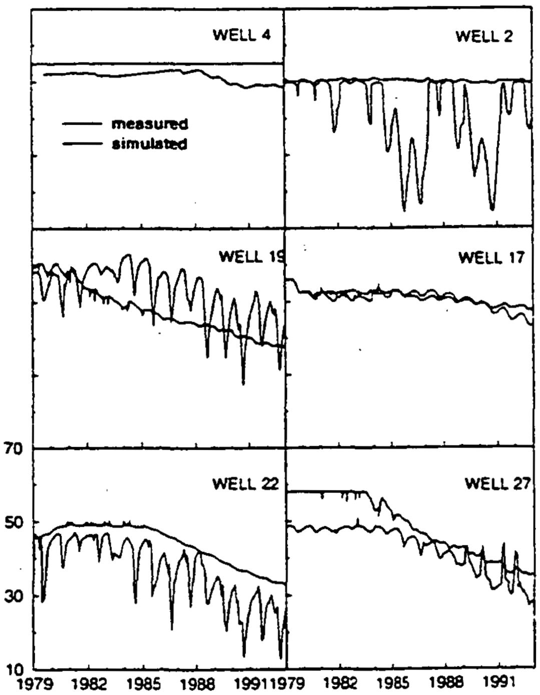


Figure 4 Comparison of measured and simulated groundwater levels

They found that it was possible to reproduce the groundwater levels with MIKE-SHE (figure 4) but not the discharge through the tunnel. Effectively the problem was that the model produced too much runoff from the hills especially in the Karstic region where there is none. Thus the tunnel discharge was overestimated.

## 3 CALIBRATION MEASUREMENTS

We were fortunate in obtaining calibration measurements for both Bells's Creek and for Lake Karla firstly through the drainage pump and secondly through groundwater records and the drainage tunnel. The groundwater level measurements in Lake Karla would not have been enough - as we have seen in Karla it is possible to simulate well the water table level without reproducing the surface water behaviour correctly. Many of the wetland functions - flooding for instance - depend more on the surface water than the groundwater. In all cases obtaining these measurements was hard and involved site visits, contacts with local agencies and uncertainties in the accuracy. In other sites there was no data at all.

In many ways calibration data are more important to modellers than input data: some input parameters, such as evapotranspiration, are unmeasureable and must be inferred indirectly whilst others have wide spatial variability or wide error bounds or both. Hydrologists therefore accept that a purely deterministic model will never be possible and results will always be obtained by adjusting one or more input parameters (hydraulic conductivity, root-depth, etc) in order to match calibration data.

We can then take advantage of the remote sensing data's great potential for model calibration. An archive of images going back at least 20 years is available over the whole of the Earth's land surface. We therefore have a period of time for which model calibration should be possible.

This has been highlighted with the cases - the North Kent marshes and Lake Karla - that we have outlined above. With this motivation in mind Shepherd et al, 1997, devised a technique for determining the water storage in a network of drainage ditches. The idea relies on the well-known principle, e.g. De Jong, 1994, that water practically absorbs all incident infra-red wavelengths (greater than $0.7\mu$) thus bands 4 ($0.76$-$0.90\mu$), 5 ($1.55$-$1.75\mu$ and 7 (($2.08$-$2.35\mu$) of the Landsat-TM sensor are suitable for searching for water.



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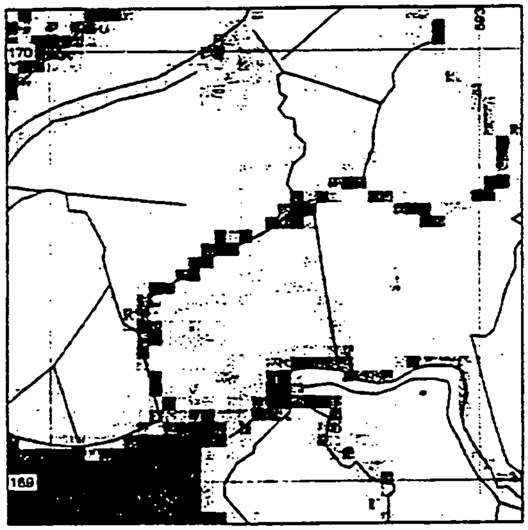



Figure 5 Ditches overlaid on band 5 of Landsat TM image (20 May 1992), image © Eurimage

The technique does not aim to give point values of water storage but rather area averages. In North Kent the ditches have an average width of 4 metres, the resolution of the Landsat-TM images is 30 metres and the technique has been tested on marshland catchments with areas varying between $1km^2$ to $10km^2$. Figure 5 shows a map of the ditches overlaid on the satellite image.

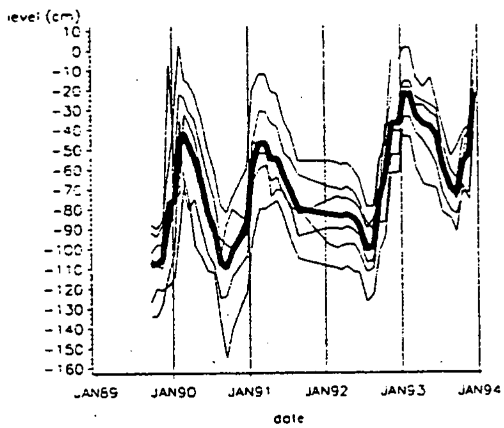



Figure 6 Measurements of ditch levels in the Elmley marshes - the thick line shows the average level

Figure 6 shows monthly measurements made in the ditches at seven points in the Elmley marshes - a specially protected zone within the marshes. These measurements have been compared at three instants with indices derived from the Landsat-TM images. A good correlation was obtained - ie when the ditches were full the indices obtained from the images was higher than when they were less full and, on the basis of this promising comparison, an article has been sent for review in an international journal.

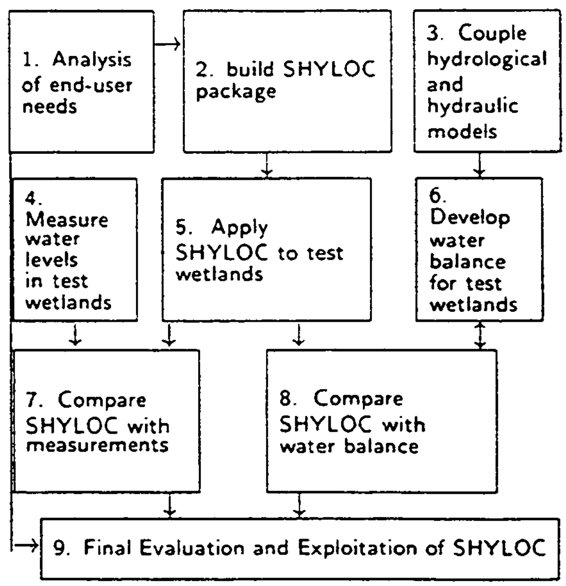
## 4 THE SHYLOC PROJECT

### *4.1 Objectives*

On the basis of these findings the SHYLOC (System for HYdrology using Land Observation for model Calibration) project started in November 1997. The primary objective of this project, funded partly by the European Commission, is to further develop both the hydrological models and the image interpretation software in order to improve the hydrological predictions and to make rigorous checks on the accuracy. A secondary objective is to make the whole process more automated, less costly in terms of manpower and capable of producing timely results.

### *4.2 Work Plan*

A flow chart of the project is shown below.




Most of the steps within the project are self-explanatory. Great emphasis is placed on involving end-users in both the specification and evaluation of the project. A number of users - English Nature, English Environment Agency, The Royal Society for the Protection of Birds, the Greek Environment Ministry, the scientific adviser to the Ramsar Bureau - have already agreed to participate. Others will be identified during the project. A supplier of earth observation data, Eurimage, is also taking an active part

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in the project in order to advise on present and future trends in environmental remote sensing.

Hydrological software is being developed by the Danish Hydraulic Institute. The MIKE-SHE model will be coupled to the MIKE-11 river package in order to improve the representation of devices such as sluices. That this coupling is feasible has already been demonstrated by Refsgaard and Sørensen, 1997, who applied a coupled model to the Danubian Lowland between Bratislava and Komárno. The objective of the present development is to put this coupling on a more operational basis.

The development of the image interpretation package is being carried out by the Joint Research Centre who will produce user-friendly automated object-oriented software integrating spatial and temporal analysis in order to produce a menu of algorithms for interpreting the images. A number of suggestions have been made for improving the accuracy of the original results and these will be implemented and tested.

The assessment of both the hydrological and image interpretation software will be carried out by teams from University College London on the English test sites - the North Kent marshes and the Pevensey Levels - and by the University of Thessaloniki as well as the Greek Biotope Wetland Centre on the lakes Karla and Mavrouda.

## 5 CONCLUSION

We have shown that deterministic modelling using commercial packages and remote sensing *remains mostly in the research domain* as far as wetland hydrology is concerned. We have shown why this is so and why the SHYLOC project offers the hope of a more operational system for hydrological predictions in the near future.


## 6 ACKNOWLEDGEMENT

This project was inspired by the late Ted Hollis who first noticed vague traces of the ditches in the satellite images and we wish that he were with us now to see the fruits of the seeds he planted. We are partially financed by the European Commission's Space Technology Programme. We thank them for their faith.


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# Geological modelling and editing in GIS environment

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ABSTRACT: Geological and hydrogeological data constitutes the basis for assessment of groundwater resources and pollution risk. The amount of available data and the heterogeneity and complexity of geological formations makes it difficult to establish a general view of the main geological features and subsequently derive a plausible geological model. This paper presents a graphical tool (GeoEditor) which provides facilities to develop and test geological models based on borehole data and geophysical data. The GeoEditor provides a close link between basic geological and geophysical data, conceptual interpretation and model representation. The GeoEditor is based on an inherent methodology, which leads the user through a systematic definition of a geological model. Based on experience of geologists, hydrogeologists and modellers two alternative approaches based on specifying either overall geological structures or zonation of characteristic aquifer properties have been implemented.

## 1 INTRODUCTION

The representation of geology is an essential part of groundwater modelling. Development of a geological model to be applied in a groundwater model is subjective depending on geological/ hydrogeological understanding, the purpose of the model (Hansen & Gravesen, 1996), the conceptualisation required for the numerical model and constraints in terms of quantity and quality of available data.

In the process of interpreting the available data into a consistent 3-D geological model one of the major problems is to visualise and combine all available data avoiding any major contradictions. It is needed to provide a general view of the overall geological structure. Consequently, a graphical tool (the GeoEditor) has been developed which links directly to a geological database (PC Zeus; Hansen, 1992) on the input side assists the selection and interpretation of field data and finally exports the geological model for further use in the hydrological modelling system MIKE SHE. The GeoEditor programme is designed to help the user in the process of establishing a conceptual model and create a geological model suited for later hydrological modelling.

*Williams et. al. (1996)* developed a GIS/AVS utility to produce geological facies based on lumping horizons of similar material properties into layer representing the accumulated layer thickness, but not the actual geological variation. The 3D geological structure was transferred to finite element grid and subsequent groundwater flow simulation. A similar GIS approach based on IDRISI and an automatic generation of multiple bedded geological surfaces is found in *Gumbricht et. al. 1996.* Other GIS tools interfacing to hydrological models, *Batelaan et. al. (1996)* are aimed at general hydrological and hydrogeological data rather than specifically geological models.

The GeoEditor is based on a different approach leaving the geological interpretation entirely to the user. Another distinguishing feature of the GeoEditor is its ability to include geo-physical data in combination with lithological borehole data which makes it highly versatile. Furthermore, emphasis is put on the post-evaluation of the validity of the numerical model.



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New survey methods provides geophysical data *Christensen et. al. (1994)* to supplement borehole data, which have traditionally formed the basis for geological interpretation. The geophysical data may not provide as unambiguous information as the borehole but the survey technique allows a vast area to be covered. With the increasing areas being covered by geophysical surveys it is thus likely that both types of data will be available in future groundwater studies. Although comparisons between the two sets of data have been presented *(Hansen & Gravesen, 1996)* a tool to access and effectively utilise the available data is needed with respect to numerical groundwater models. The GeoEditor presented in the following reads the geological input data from the PC Zeus database and has been interfaced to the MIKE SHE hydrological modelling system.

This paper presents a graphical geological editor developed in GIS environment (ArcView from ESRI) which allows the user to:

- Select a subset of borehole data and geophysical data.
- To visually interpret the data in order to establish a geological model.
- To utilise other types of digital point and area data such as geological and topographical maps.
- To transfer surfaces of geological units to a numerical model simulating groundwater flow and solute transport.
- To validate the discretisized model against all available geological data.

Hydrogeological parameters must be specified to set-up a groundwater model. Spatially and temporally distributed hydrogeological field data may be introduced in the GeoEditor to support the task.

## 2 APPROACH

A three-phase approach has been adopted where selection of data is followed by geological interpretation. The overall geological structure may be specified in terms of layers and lenses by stepwise sweeping through a number of predefined geological profiles or alternatively by specifying zones of characteristic aquifer types sequentially for a number of depth intervals from the ground surface to the deep aquifers. In the third step the discrete values are interpolated into a 3-dimensional geological model. The resulting geological model is, subsequer evaluated and modified if rejected. When accepted it is transferred to input files of the groundwater simulation model. In the design of the geological editor the graphical element is essential. Consequently the initial selection of borehole data has been designed for visual selection from the PC Zeus database (national borehole database of Denmark) and reduction of the sub-database by various selection criteria.

In the geological interpretation of data two alternative methods are available. A series of cross-sectional views are provided, which allows the user to connect discrete lithological data into a continuos model based on assumptions of overall geological structures. Alternatively the user may sweep through a series of horizontal slices. The slice represents a certain de interval and the proportionally dominant lithological type is shown. The user must draw up zones which share the same hydrogeological characteristics. The number of lithological classes determines the detail of the model. The latter approach is useful in a highly heterogeneous geology.

In the final phase the geological model may be evaluated by means of 3-D graphics and statistical methods. The geological model is transferred into the computational grid of a model and it is checked to which extent the model representation fits the geology of the area. Hydrogeological data may be retrieved to support the geological interpretation and to estimate parameters to be entered for geological layers in the model. Distribution of transmissivities and storage coefficients is a typical example of relevant field data which may be useful for the conceptual geological model.

Cross checking with screened abstraction wells and observation wells gives an indication of the accuracy of the geological model. Filters overlapping high and low yielding layers of the geological model may be attributed to erroneous or incomplete geological specifications. Typically it will take a few iterations before a satisfactory geological model has been derived or alternatively a number of possible geological hypothesis may be tested. The geological information is transferred to input files for the numerical model.



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## 3 SELECTION OF DATA

Initial selection of borehole information

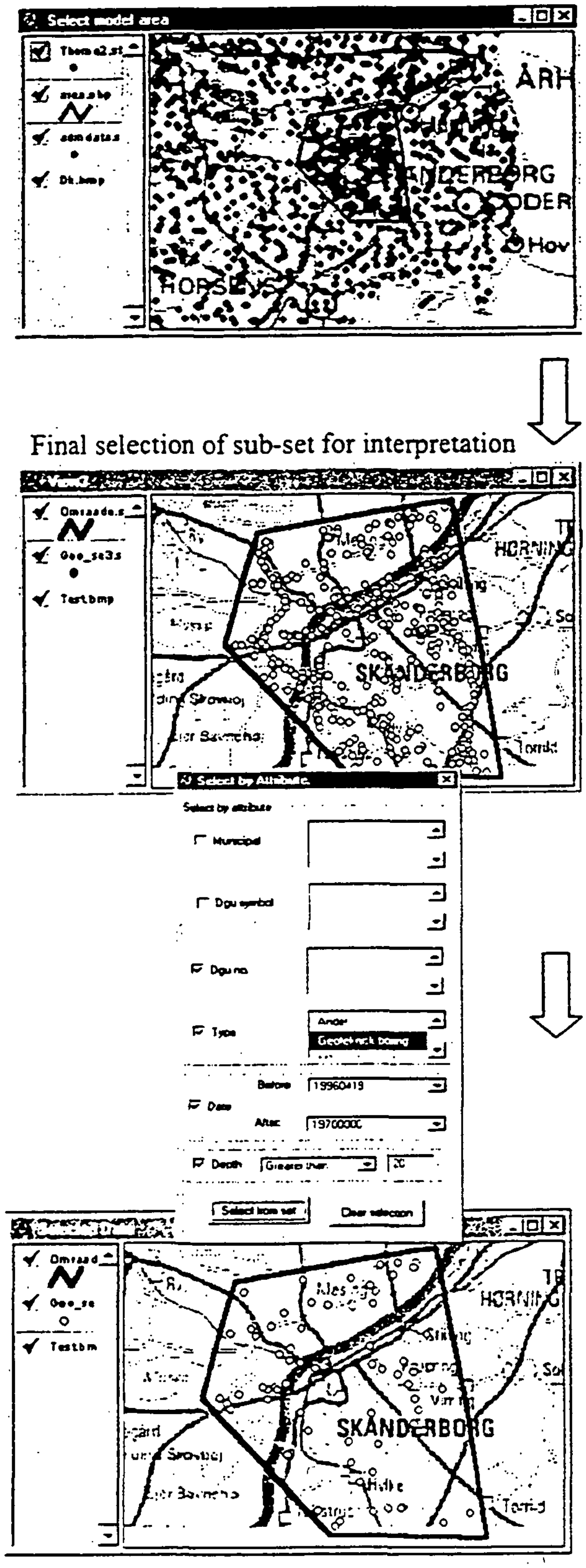


Figure 1 Selection of borehole data

A sub-set of data is selected to reduce the density of data and to exclude data of lesser quality. The geographical area of interest is specified on a base map and the data within the area are selected. Subsequently further reduction of the data may be needed. Selection criteria for borehole age, purpose of the borehole and the actual lithological types are included. Single boreholes may be excluded if necessary. It is also possible for the user to type in additional information's either as new borings or as ghost data (guiding points in area with sparse information). The new data are stored and displayed separately to hinder a mixture of the different data types.

## 4 GEO-PHYSICAL DATA

Geo-physical data provides soil resistance of the geological layers. The range of resistance reflects the porosity or matrix properties of the layers and the water content of the soil. Under saturated conditions specific sediment or rock types can be associated with approximate intervals.

By adopting a relationship between restivity and lithological classification comparison between the two sets of data is possible.

Table 1 Approximate correlation between restivity and sediment type

| Restivity (Ω) | Soil type | Lithological type |
|---|---|---|
| 1 - 20 | Clay | l |
| 20 - 70 | Clay loam | ml |
| 40 – 100 | Sand loam | ms |
| 60 - 200 | Limestone | k |
| 101 - 200 | Sand (wet) | ds |
| 201 - 10000 | Sand (dry) | s |

Precaution should be taken when applying the table for comparison with lithological types found from boreholes. The contrasts in restivity are not very distinct, the intervals are in fact overlapping and the signals may be distorted by electrical installations in the field. In series of interbedded layers it is difficult precisely to identify the transition between sediment types. Consequently geo-physical data alone are often not adequate to establish a geological model. On the other hand techniques have been developed to cover large areas by geo-physical surveys and geo-physical data may be an important source of information in areas with scarce borehole data.

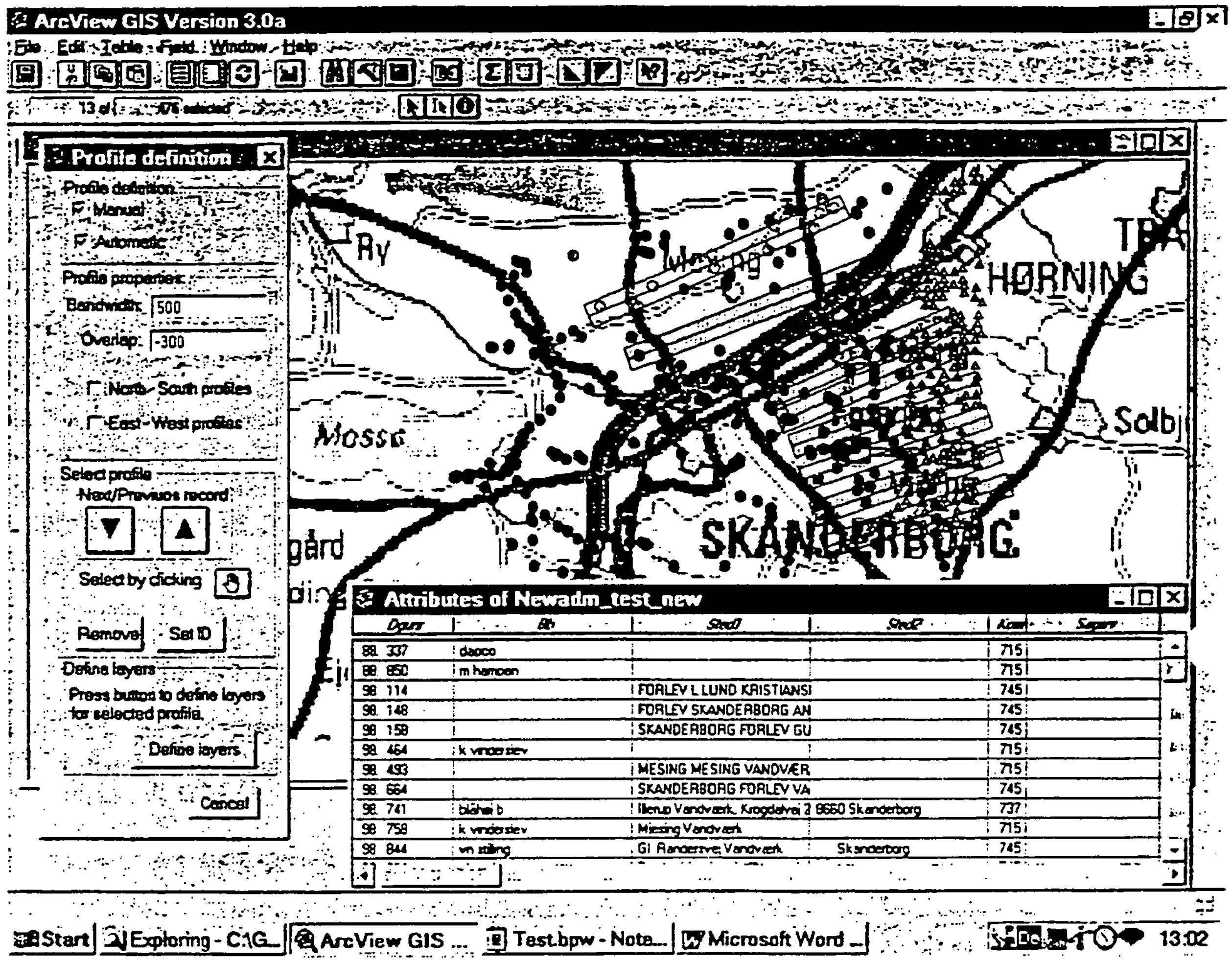


Figure 2 Location of geological profiles

The penetration depth of geo-physical surveys is up to 150 m, which is significantly deeper than most boreholes. Where boreholes are not sufficiently deep geo-physical data are applied.

Borehole data are treated as the primary source of information and geo-physical data serve as 'background' information. The interpretation of geo-physical data and their correlation to borehole information need further development and refinement.

## 5 GEOLOGICAL MODEL

Upon selection of a sub-set of borehole data and geo-physical data the geological model is established. The interpretation requires a degree of conceptualisation which must correspond to the purpose of the model. The scale of geological variability and the horisontal and vertical discretisation of the numerical model must be considered. Based on the physical understanding a number of geological units or alternatively characteristic lithological classes must be specified depending on the preferred method of interpretation.

Geological cross sections are specified with projected boreholes and a background colour f-- geophysical data. The profiles may be inspe to arrive at a reasonable number of geological units.

The geological model is established by moving through the profiles connecting each geological layer or lense from borehole to borehole. To maintain an overview during the process horisontal and vertical views are combined.

Horizontal coordinates of the borehole and the elevation are saved in all connection points of all profiles for each geological unit.

Other types of data, e.g. topographical profiles or layer profiles of previous geological models may be added as external sources of information to support the interpretation.

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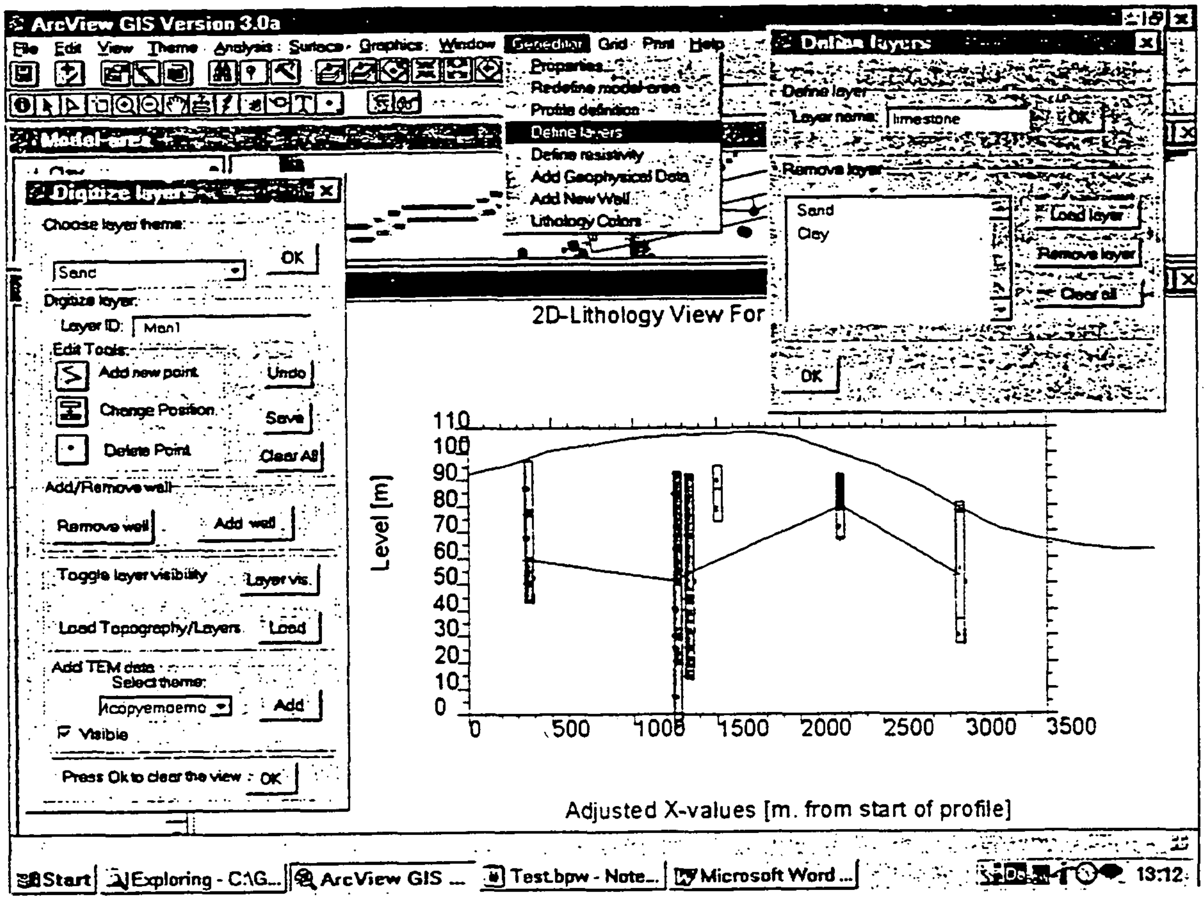


Figure 3 Description of layers in profiles

## 6 CONVERSION TO MODEL INPUT

Coordinates of the points connecting the geological layers are interpolated into 3-D surfaces. A grid corresponding to the spatial discretization of the model is adopted and files describing top and bottom of each geological unit are interpolated from discrete data points. Output is generated in terms of generalised grid maps – the format has been chosen to comply with the MIKE SHE model.

## 7 MODEL EVALUATION

Spatial discretization is needed to transfer the geological layers to a numerical model. Consequently the geological representation of the numerical model may deviate from the user specified geological model. Due to limitations regarding the computer capacity a coarse numerical grid may have to be applied and spatial

resolution causes details to be lost or distorted. It is thus of great importance to check if the conceptual understanding inherent in the geological model is actually present in the model representation. By overlaying surfaces of the geological model and the gridded maps to be used in the numerical model on top of the basic physical data, it is possible to point out differences attributed to the conversion between the continuous and discrete domain. Modifications of the numerical grid resolution or the geological layer geometry may be necessary to obtain a satisfactory model representation.

A unique geological model can in most cases not be obtained. Instead different geological hypothesis may be investigated with respect to the effect on simulated potential head and groundwater flow. Testing different options may serve to exclude or include geological units within the model area.

## 8 CONCLUSIONS

The GeoEditor is designed to support the pre-processing of geological data, taken directly from the PC Zeus database, and the

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establishment of a geological model suited for hydrological modelling MIKE SHE). The primary objective is an improved representation of major geological features in a groundwater model. The GeoEditor can be seen as an advanced and flexible pre-processing utility for groundwater modelling.
Automatic generation of the geological model has contrary to other GIS based geological tools not been attempted. Due to the geological complexity and the conseptualisation required it is not feasible to the general understanding and knowledge of the user. A geological model is by no means unique and must reflect the intended use. The GeoEditor facilitates the process by providing a graphical environment with an easy access to basic data, but leaves it entirely to the user to develop a plausible geological model.

The benefit of applying both borehole data and geophysical data is not yet fully investigated but work by Hansen & Gravesen (1996) suggest that especially the deeper parts of the model will benefit from this additional data type. The collocation of different types of data provides a more complete picture by filling out gaps found for the respective data types. The conversion between measured geo-physical data and the lithological classification is critical and to some extent uncertain in heterogeneous sedimentary deposits. The inclusion of geo-physical data will however substantially improve the geological models.

The practical experience applying the GeoEditor is limited so far. Even though the GeoEditor is developed to facilitate development and test of geological models to be used in hydrological models and groundwater models it clearly has a potential to improve descriptive geological models in general.

The geological model, the hydrogeological parameters and the boundary conditions affect the performance of the numerical model. Although the GeoEditor focuses on the geological lay out the GIS framework is suited for integrated analysis including parameterisation and boundary conditions.

Attention should be paid to a numerical models sensitivity with respect to parameters and other input data such as the geological mode. The GeoEditor supports the investigation of various geological hypothesis and thereby quantify the uncertainty referring to the geological model.

Further development of the GeoEditor is aimed towards establishing a closer link between numerical model and GIS pre-processing facilities in order to obtain an integr modelling utility.

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# Uncertainty in spatial decision support systems – Methodology related to prediction of groundwater pollution

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ABSTRACT: Groundwater pollution from non point sources, such as nitrogen from agricultural activities, is a problem of increasing concern. This paper presents a methodology and a case study for large scale simulation of aquifer contamination due to nitrogen leaching. Comprehensive modelling tools of the physically-based type are well proven for small scale applications such as plots or small experimental catchments with good data availability. The two key problems related to large scale simulation are data availability at the large scale and model upscaling to represent conditions at larger scale. In this study readily available data from standard European level data bases such as GISCO and EUROSTAT have been used as the basis of modelling. These data were supplemented by selected readily available national data sources. The model parameters were all assessed from these data by use various transfer functions, and no model calibration was carried out. Furthermore, a statistically based upscaling/aggregation procedure, preserving the areal distribution of soil types, vegetation types etc on a catchment basis has been adopted. Finally, a Monte Carlo simulation technique was used to assess how uncertainty in selected input data propagate through the model and results in uncertainty on the model outputs. The case study from the Karup catchment in Denmark indicate that the resulting uncertainty of the predicted nitrogen concentrations in the aquifer at a scale of some hundreds of $km^2$ is so relatively small that the methodology appears suitable for large scale policy studies.

## 1 INTRODUCTION

Groundwater is a significant source of freshwater used by industry, agriculture and domestic users. However, increasing demand for water, increasing use of pesticides and fertilisers as well as atmospheric deposition constitute a threat to the quality of groundwater. The use of fertilisers and manure leads to the leaching of nitrates into the groundwater and also contributes to the acidification of soils that can also have an indirect effect on the contamination of water.

In Europe, for instance, the present situation is summarised in EEA (1995), where it is assessed that the major part of aquifers in Northern and Central Europe are subject to risk of nitrogen contamination due to agricultural activities. Therefore, policy makers and legislators in EU are concerned about the issue and a number of preventive legislation steps are being taken in these years (EU Council of Ministers, 1991; EC, 1996).

Most nitrogen leaching models such as RZWQM (DeCoursey et al., 1989; DeCoursey et al., 1992) and DAISY (Hansen et al., 1991) are of the physically-based type, but cover only the root zone at plot or field scale. Traditionally, complex leaching models are only used on plot or field scales in areas with extraordinarily good data availability, and even for such cases the relevance of such approach is often questioned because of the perceived uncertainty related to the model simulations. Hence, there is an evident need to assess the uncertainty related to large scale simulation of aquifer pollution from diffuse sources.

To our knowledge, so far no attempts to assess the model uncertainties of simulated groundwater contamination due to nitrogen leaching based on a true, spatially distributed catchment model using physically-based process descriptions have been reported. The present paper addresses this issue with special emphasis on large scale modelling.



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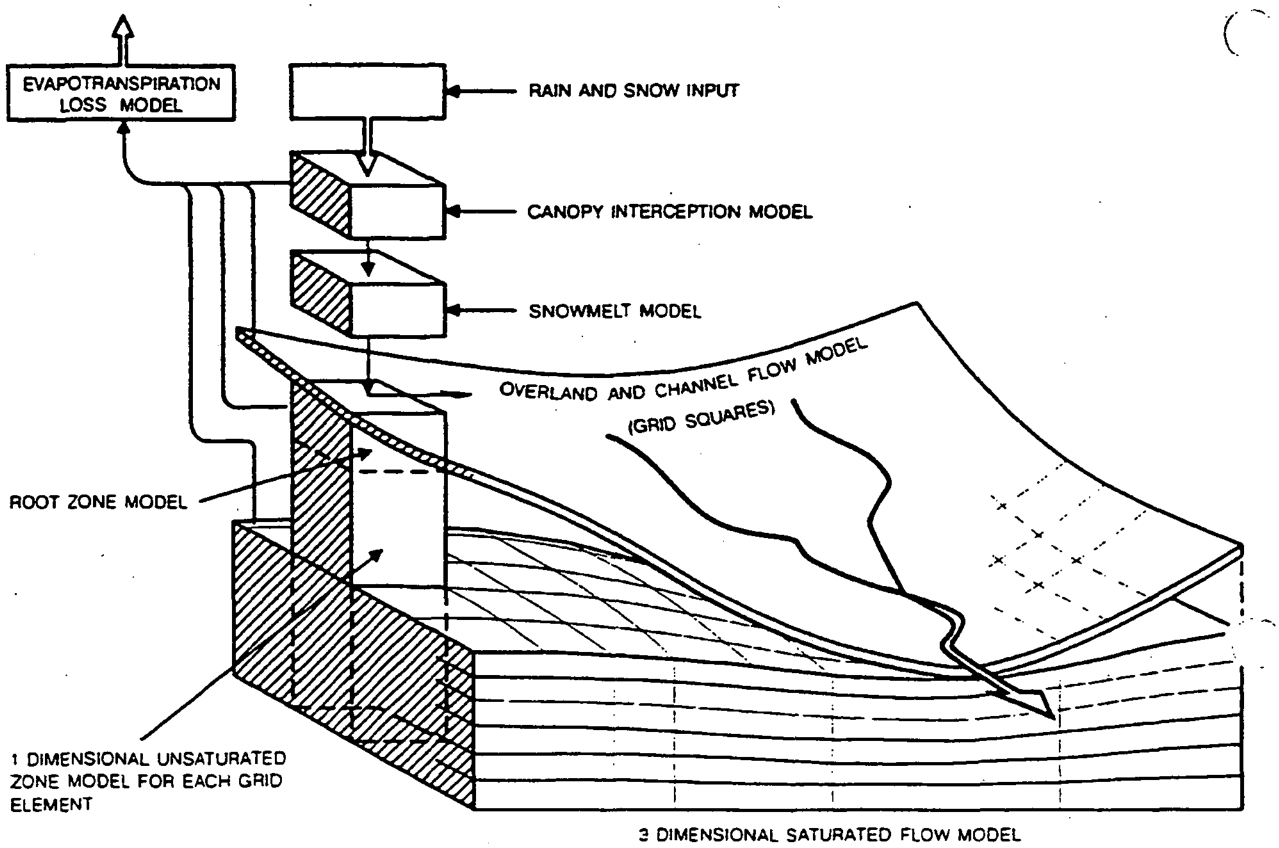


Figure 1 Schematic representation of the components of MIKE SHE

## 2. METHODOLOGY

### 2.1 Deterministic simulation models

*MIKE SHE*

MIKE SHE is a modelling system describing the flow of water and solutes in a catchment in a distributed physically-based way. This implies numerical solutions of the coupled partial differential equations for overland (2D) and channel flow (1D), unsaturated flow (1D) and saturated flow (3D) together with a description of evapotranspiration and snowmelt processes. The model structure is illustrated in Fig. 1. For further details reference is made to the literature (Abbott et al., 1986; Refsgaard and Storm, 1995).

*DAISY*

DAISY (Hansen et al., 1991) is a one-dimensional physically based modelling tool for the simulation of crop production and water and nitrogen balance in the root zone. DAISY includes modules for description of evapotranspiration, soil water dynamics based on Richards' equation, water uptake b plants, soil temperature, soil mineral nitrogen dy namics based on the advection-dispersion equatio nitrogen uptake by plants and nitrogen transform tions in the soil. The nitrogen transformatior simulated by DAISY are mineraliz immobilization turnover, nitrification and de cation. In addition DAISY includes a module f description of agricultural management practices.

*MIKE SHE/DAISY coupling*

By combining MIKE SHE and DAISY, a comple modelling system is available for the simulation water and nitrate transport in an entire catchment. the present case the coupling is a sequential on Thus for all agricultural areas, DAISY first produc calculations of water and nitrogen behaviour fro soil surface and through the root zone. The percol tion of water and nitrogen at the bottom of the ro zone simulated by DAISY, is then used as input MIKE SHE calculations for the remaining part the catchment. For natural areas, MIKE SHE ca culates also the root zone processes assum

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nitrogen contribution form these areas. Due to the sequential execution of the two codes, it has to be assumed that there is no feed back from the groundwater zone (MIKE SHE) to the root zone (DAISY).

Thus, MIKE SHE does not in the present case handle evapotranspiration and other root zone processes in the agricultural areas. As DAISY is one-dimensional, one DAISY run in principle should to be carried out for each of MIKE SHE's horizontal grids. However, several MIKE SHE grids are assumed to have identical root zone properties (soil, crop, agricultural management practices, etc), so that in practise the outputs from each DAISY run can be used as input to several MIKE SHE grids.

### 2.2 *Upscaling of models for application to large scale*

Large scale hydrological models are required for a variety of applications in hydrological, environmental and land surface-atmosphere studies, both for research and for day to day water resources management purposes. The complex interaction between spatial scale and spatial variability is widely perceived as a substantial obstacle to progress in this respect (Blöschl and Sivapalan, 1995; and many others).

The basic upscaling methodology adopted here is in accordance with the principle recommended by Heuvelink and Pebesma (submitted), as follows:

- The basic modelling system is of the distributed and physically-based type. For application at *point scale* (where it is not used spatially distributed) the process descriptions of this model type can be tested directly against field data.
- The model is in this case run with (equations and) parameter values in each horizontal grid point representing *microscale* conditions. The microscale is selected as a scale for which previous tests have indicated that the model is able to describe the basic processes satisfactorily (50 - 200 m).
- The smallest horizontal discretization in the model is the *grid scale* or grid size (1 – 5 km) that is larger than the microscale. This implies that all the variations between categories of soil type and crop type within the area of each grid can not be resolved and described at the grid level. Input data that vary at microscale, and whose variations are not included in the grid scale model representation, are distributed at the macroscale so that their statistical distributions are preserved at that scale.
- The results from the microscale modelling are then aggregated to *macroscale* (10 – 50 km) and the statistical properties of model output and field data are then compared at macroscale.
- For applications to larger scales than macroscale, such as European or *continental scale*, the macroscale concept is used, just with more grid points. This implies that the continentel scale can be considered to consist of several macroscale units, within each of which the microscale statistical variations are preserved and at which scale the predictive capability of the model thus lies.

It should be emphasised that the microscale considered is not point scale, but rather a field scale characterised by 'effective' soil and vegetation parameters, but assuming only one soil type and one cropping pattern. Thus the spatial variability within a typical field is aggregated and accounted for in the 'effective' parameter values. Physically-based models like MIKE SHE and DAISY have previously at many occasions documented their ability to describe conditions at field scale by use of effective parameters (e.g. Jensen and Refsgaard, 1991a,b,c). In the aggregation to macroscale the variations among soil types and crops are preserved statistically, although they are not correctly georeferenced.

### 2.3 *Modelling approach*

In large scale modelling the aim is to simulate processes, in this case the nitrogen contamination of aquifers, at an areal scale larger than point or field scale. This implies that the model outputs do not pretend to reproduce point values and as such can not directly be compared to point observations from e.g. wells. The model results should rather be compared to field data in terms of statistical properties within a given area. Hence, the modelling aims at describing the integrated runoff at the catchment outlet and the distribution function of the nitrogen concentrations sampled from available wells over the catchment (aquifer).

In this study readily available data from standard European level data bases such as GISCO and EUROSTAT have been used as the basis of modelling. These data were supplemented by selected readily available national data sources. The model parameters were all assessed from these data by use various transfer functions, and no model calibration was carried out.

### 2.4 *Uncertainty analyses*

The uncertainty analyses is based on a Monte Carlo

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simulation technique with an efficient stratified sampling approach (Pebesma and Heuvelink, in preparation).

There are uncertainties related to all input data. For practical reasons the number of parameters for which uncertainties could be considered in the Monte Carlo analyses had to be limited. Therefore, the following five parameters which were known by experience to be the most dominant and the most uncertain ones in relation to prediction of nitrogen contamination were selected:

- Precipitation. The error on the daily precipitation was assumed to follow a normal distribution with zero mean and a standard deviation equivalent to 50% of the measured daily value. This value is in good agreement with field data (Allerup et al., 1982)
- Soil hydraulic properties. The hydraulic conductivity and soil moisture retention curves were estimated from the textural composition by use of standard pedo-transfer functions, based on Cosby et al (1984). The clay content was assumed uniformly distributed within the range given by the texture class in the FAO classification used in the GISCO data base. and the relation between clay, silt and sand were fixed by an empirical equation.
- Soil organic matter. The uncertainty on this parameter was assessed from experience and assumed to follow a truncated normal distribution with a relatively high coefficient of variation of 0.45.
- Slurry composition. The uncertainty in dry matter and total nitrogen content in the slurry was assessed from experimental data (T. Bonde. personal communication). The coefficients of variation of the various slurry types vary from 0.25 to 0.50.
- Depth of reduction front in the aquifer. This was assumed to vary within a range of 9 m.

As the CPU time requirements for a model simulation with the deterministic integrated MIKE SHE/DAISY model in itself is substantial. it was required to keep the number of Monte Carlo runs at a minimum. In this case 25 runs were carried out and comparisons with additional 50 runs indicated that 25 runs were sufficient to assess the uncertainty on the mean values.

## 3. CASE STUDY

### *3.1 Area and model construction*

The area used in the study is the Karup river basin which is located in the middle part of Jutland, Den mark (Figure 2). The topographic catchment c approximately 500 km$^2$ of which 70 % are used for agricultural purposes and 30 % are natural areas The catchment characteristics are described in Styczen and Storm (1993). The data used for the present study and the model construction are de scribed in details in Refsgaard et al. (in preparation) In the following a brief summary is provided.

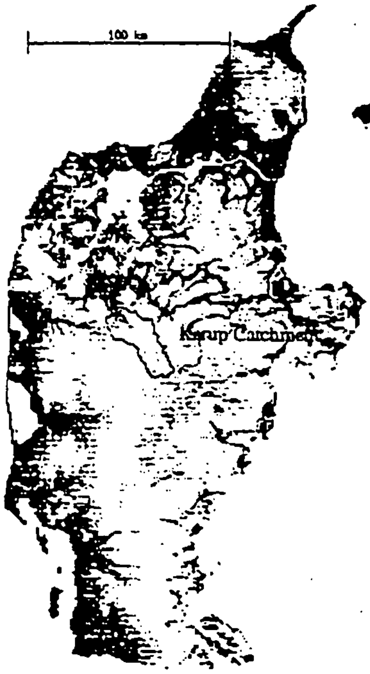


Figure 2 Location of the Karup catchment in Jutland Denmark

The catchment was in the model represented three-dimensional network. The discretization used for the uncertainty analysis was 2 km in the horizontal directions and varied in the vertical from 5 to 40 cm in the unsaturated zone. and from 10 to 15 m in the satu rated zone. Catchment delineation and stream geome try was carried out on the basis of a digital elevation map from USGS/GISCO (DEM) using Arc/Info fa cilities. Spatial distributions of land use and soil types were derived from the GISCO database and hydro geological data were obtained from EC (1982). Dis tributions of crop types and livestock densities were obtained from Agricultural Statistics (1995) and converted to slurry production using standard values for Nitrogen content. Based on typical crop rotations proposed by The Danish Agricultural Advisory Centre and the constraints offered by crop distribu tion and livestock density two cattle farm rotat one pig farm rotation and one arable farm rotation were constructed. In order to capture the effect of the interaction between weather conditions and crop, simulations were performed in such a way that each crop at its particular position in the considered rotation occurred exactly once in each of the years, which resulted in a total of 17 crop rotation schemes. These 17 schemes were distributed randomly over the area in such a way that the statistical distribution was in accordance with the agricultural statistics.

To simulate the trend in the nitrate concentrations in the groundwater and in the streams, it is necessary to have information on the history of the fertiliser application in space and time. In Denmark, norms and regulations for fertilisation practice are defined (Plantedirektoratet, 1996) which regulate the maximum amount of nutrients allowed for a particular crop depending on forefruit and soil type, and in addition, provide norms for the lower limit of nitrogen utilisation for organic fertilisers. It was assumed that the farmers follow the statuary norms, and that the proportion of organic fertiliser to the individual crop in a rotation is proportional to the production of organic fertiliser in the rotation and to the relative nitrogen demand of the crop (the fertiliser norm of the particular crop in relation to the fertiliser norm of the rotation). Based on estimated application rates of organic and mineral fertiliser to the individual crops each year, the DAISY model simulated time series of nitrate leaching from the root zone for each agricultural grid. The MIKE SHE model then routed these fluxes further through the unsaturated zone and in the groundwater layers accounting for dispersion and dilution processes and finally into the Karup stream where the integrated load from the entire catchment was estimated. The model was run for seven years (1987–93). In order to reduce the importance of uncertain initial conditions, the two first years were considered as a 'warming-up period' and the last five years were considered the simulation period. As annual leaching depends on weather, crop and crop position in the rotation, groundwater concentrations in single years were not considered, instead data averaged over the five year simulation period, 1989-93, were used for the uncertainty analysis.

### 3.2 Results of deterministic model

The simulated discharge hydrograph is shown in Figure 3 together with the observed one. Considering that the model is very coarse and uncalibrated it reproduces measured river runoff relatively good. In particular the long term annual average runoff is remarkably well reproduced by the model. On the other hand, the distribution of the river runoff over the year is not so well reproduced. The model generally underestimates the runoff in the summer periods (low flows) and overestimates the winter flow.

Figure 4 shows the simulated nitrate concentrations in the upper ground water layer compared to observed values from 35 wells in terms of statistical distribution over the aquifer. The simulated concentrations are seen to match the observed data surprisingly well.

### 3.3 Results of uncertainty analyses

The key results from the Monte Carlo runs are shown in Table 1. The figures in the table are averaged over the five year simulation period, and the mean and standard deviations represent statistical values from the 25 Monte Carlo runs. The uncertainties on the simulated average groundwater concentrations are further shown in Figure 5.

Table 1 Key results from 25 Monte Carlo runs

| Variable | Mean | St.dev. |
|---|---|---|
| Leaching from the root zone (kg/ha/year) | 65 | 19 |
| Groundwater concentration (mg/l) | 48 | 8 |
| River flow (mm/year) | 464 | 22 |

## 4 DISCUSSION AND CONCLUSIONS

From the results of the present study it can be concluded that the present modelling approach appear feasible for estimating uncertainties in predicted nitrate concentrations at larger scales, and hereby also for evaluating the reliability of the simulation results. Results also show that even though the uncertainty of the simulated results is large at point/grid scale, making the predictive capabilities questionable, it appears that the uncertainty at larger scales where the point simulations are integrated in time and space become much smaller and hereby appear more useful.

Given the rather coarse data basis, and given that no model calibration was performed, the simulated results of groundwater concentrations were remarkably good. Hence the measured groundwater concentrations fell well within the predicted uncertainty intervals.

Thus it appears that the methodology could relatively easy be applied to larger areas and used as decision support tools for evaluation of legislative measures aiming to reduce nitrogen contamination risk.

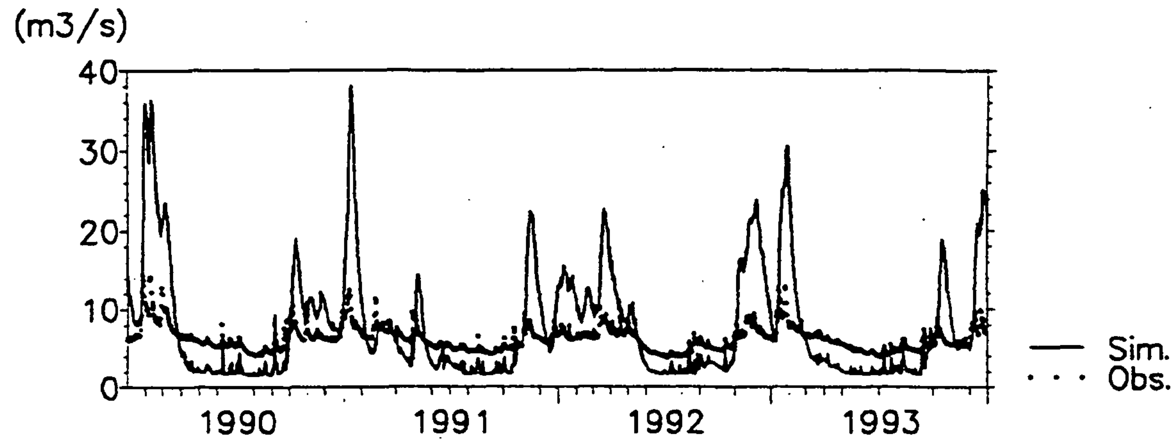


Figure 3 Simulated and recorded discharge hydrographs for the Karup catchment 1990-93

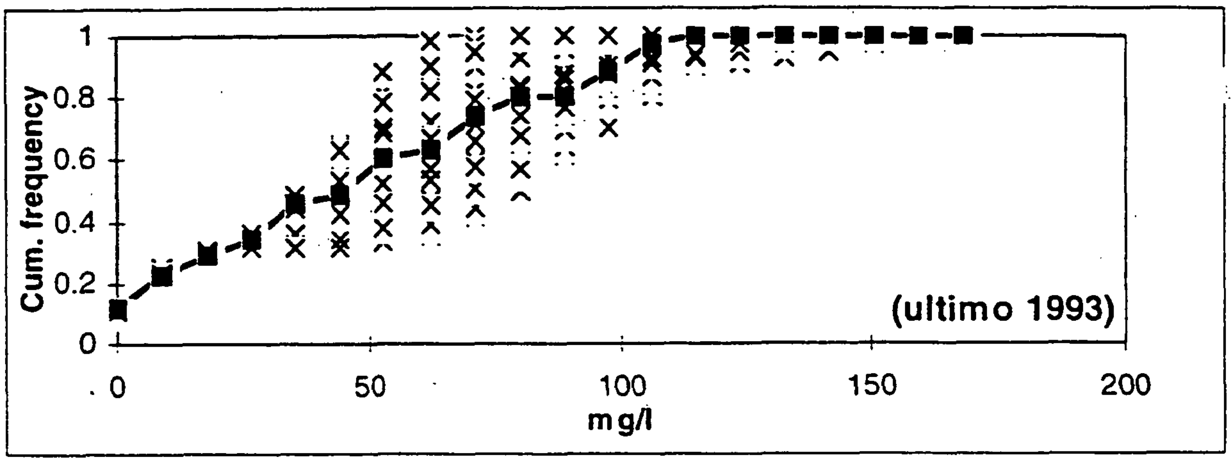


Figure 4 Statistical distribution of groundwater concentrations over the Karup catchment by the end of 1993. The fu line with squares corresponds to the measured data while the lines with x are the 25 Monte Carlo runs.

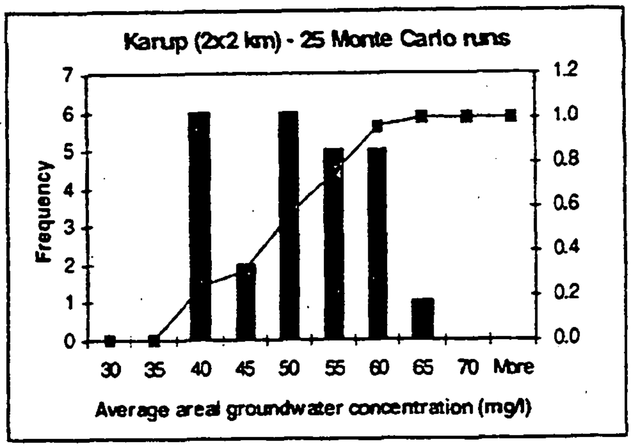


Figure 5 Statistical distribution over 25 Monte Carlo runs of simulated areal average $NO_3$

## 5 ACKNOWLEDGEMENT

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# Assessment and minimisation of environmental impacts of a planned reservoir using an integrated modelling approach

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## Introduction

This paper presents the results of applying an integrated model complex in the planning and design phase of a combined drinking water, flood control and recreational reservoir in the Prosna River, Poland, named the Wielowies Reservoir.

The main purposes of the reservoir, with an estimated 50 mill. $m^3$ water storage capacity, are to improve the possibilities for flood control, to secure a water supply for irrigation, to provide the cities Kalisz and Ostrów with drinking water and the potential of future hydropower production.

A combined hydraulic and water quality one-dimensional model has been applied in the river branches, upstream and downstream the proposed reservoir, whereas a two-dimensional vertical model description has been used in the reservoir part of the system. The main purpose of the river part of the model, apart from provision of boundary data for the reservoir model, is to assess flows, water levels, sediment transport and water quality affected by the construction of the reservoir in reaches along the Prosna River. The oxygen conditions, re-cycling of nutrients (nitrogen, phosphorus), degradation of organic matter, as well as the biological parameters: phytoplankton biomass and chlorophyll-a, have been taken into account in the water quality studies in both parts of the system: the river branches and the reservoir.

A GIS-based river basin model has been used to simulate water distribution and utilisation within the catchment area, as well as reservoir outflows from various rule of operation curves. Groundwater infiltration from the reservoir has been simulated via a fully three-dimensional groundwater model.

The assessment and optimisation of different alternative designs utilising the integrated model systems demonstrated that the storage capacity of the reservoir can be made sufficient for the design flood provided that design flood control levels are not exceeded. Sediment transport simulations demonstrated that siltation will be slow and not affect the lifetime of the reservoir. Downstream erosion combined with backwater sedimentation will, however, require regular maintenance work.

Due to the high nutrient load the oxygen conditions in the reservoir will be poor and anaerobic conditions will occur during the summer. Modelling of the reservoir processes demonstrated that even a 50% reduction of the nutrient load from the upstream sources will not eliminate the eutrophication problems of the reservoir, including the oxygen conditions. A reduction of more than 50% of the nutrient load is necessary in order to obtain a satisfactory environmental quality throughout the reservoir.

Infiltration of reservoir water and the increase in head had only minor impacts on groundwater flow and quality in adjacent areas as verified by the coupled groundwater model.

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## 1. Background

The need for freshwater supply for various purposes is an increasing problem in many countries in Eastern Europe, including Poland. Today, in Poland, the water demand for irrigation and drinking water supply is increasing and the demand will be even more intense in the years to come. In order to meet the increasing need for water supply the Polish Government has planned to build a number of reservoirs throughout the country up to year 2005. The construction of reservoirs is part of a national plan for flood control and water resource management.

The planned combined drinking water, flood control and recreational reservoir near Wielowies Klasztorna in the Prosna River Basin (Figure 1) is an example of a reservoir included in the above mentioned nation-wide plan. The reservoir is intended to improve the possibilities for river flow regulation and flood control, to secure water supply for irrigation and, in the future, to provide drinking water to the cities of Kalisz and Ostrów Wielkopolski. Finally, the reservoir will provide recreational possibilities for the local population in the region.

The Prosna River joining the Warta River, the third largest river in Poland, is 225 km long and has a catchment area of almost 5000 km$^2$. The Wielowies Klasztorna Reservoir will be located in the central part of the river reach where typical peak flows reach 80 m$^3$/s and average summer flows are in the order of 5 m$^3$/s. The present water quality conditions in the river are strongly affected by point sources and diffuse agricultural loads along the river /6/.

In Figure 1 the location of the proposed Wielowies Klasztorna reservoir is shown.

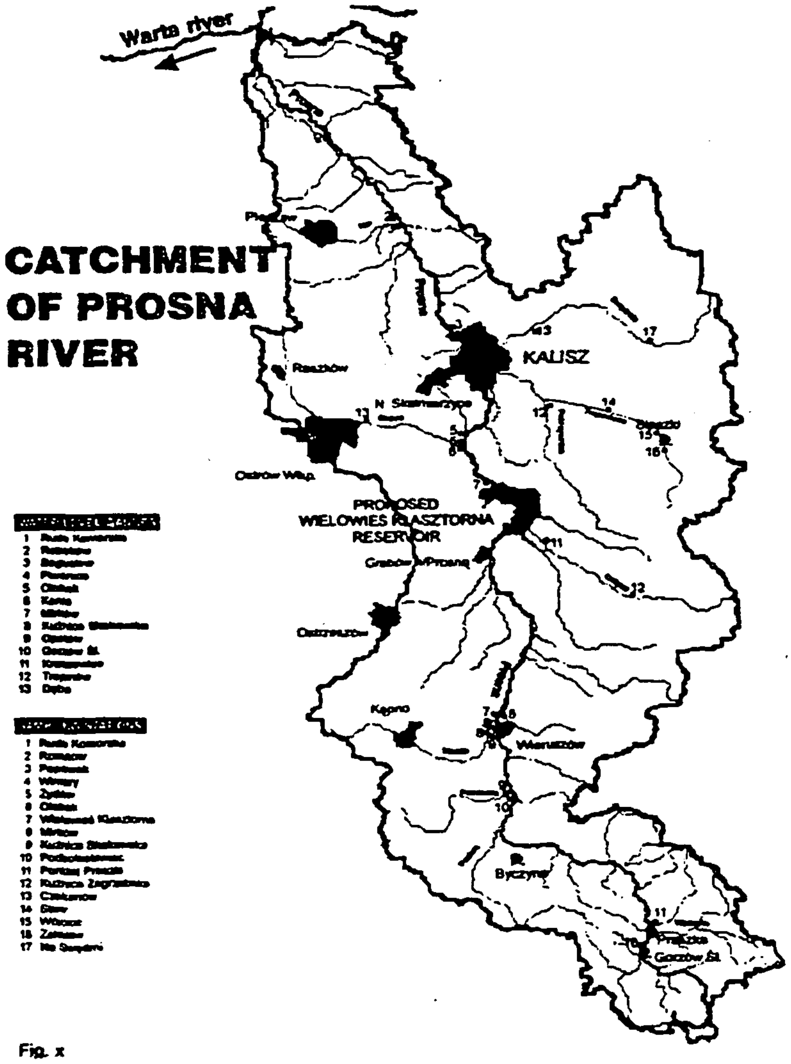


Fig. x

Figure 1. Study area and location of the reservoir.

The construction of Wielowies Klasztorna reservoir will inevitably affect the natural conditions with respect to surface and groundwater flow and water quality and hence the flora and fauna in Prosna River and adjacent areas. Several environmental problems in connection with the construction are foreseen. Especially eutrophication (nutrient enrichment), siltation and poor water quality in and downstream the reservoir could be critical with regard to the future aim of the reservoir. Changes in groundwater levels and groundwater quality in the adjacent areas are also foreseen.

In order to study the combined impacts from catchment inflow and operational strategies on the physical and biological processes within the reservoir as well as in downstream and upstream affected reaches, the Polish Regional Water Management Board requested a comprehensive integrated modelling system. The modelling system should be so general that it can be applied to the design of the Wielowies Reservoir and also to future reservoir designs and constructions expected within the next decade in Poland.

## 2. Methods

Effective management and operation of a multi purpose water body such as the Wielowies Klasztorna Reservoir has to be based on an understanding of the related physical and chemical/biological processes. Obviously the impact of the reservoir on the surrounding environment and vice versa requires an understanding of the processes in the entire river basin.

A number of models exist tailored in particular to modelling the individual processes. The integration of models and results often proves to be costly and time consuming making the models inefficient and unpractical as a management tool and during the reservoir design phase. However, combining models with a flexible structure sharing a common platform for data pre- and post processing can provide an efficient tool in connection with design and implementation of water management systems in general. Such a tool has been applied to the design and optimisation of the Wielowies Reservoir. The processes, modular structure and data flow of the integrated model complex are illustrated in Figure 2 and Figure 3, respectively.

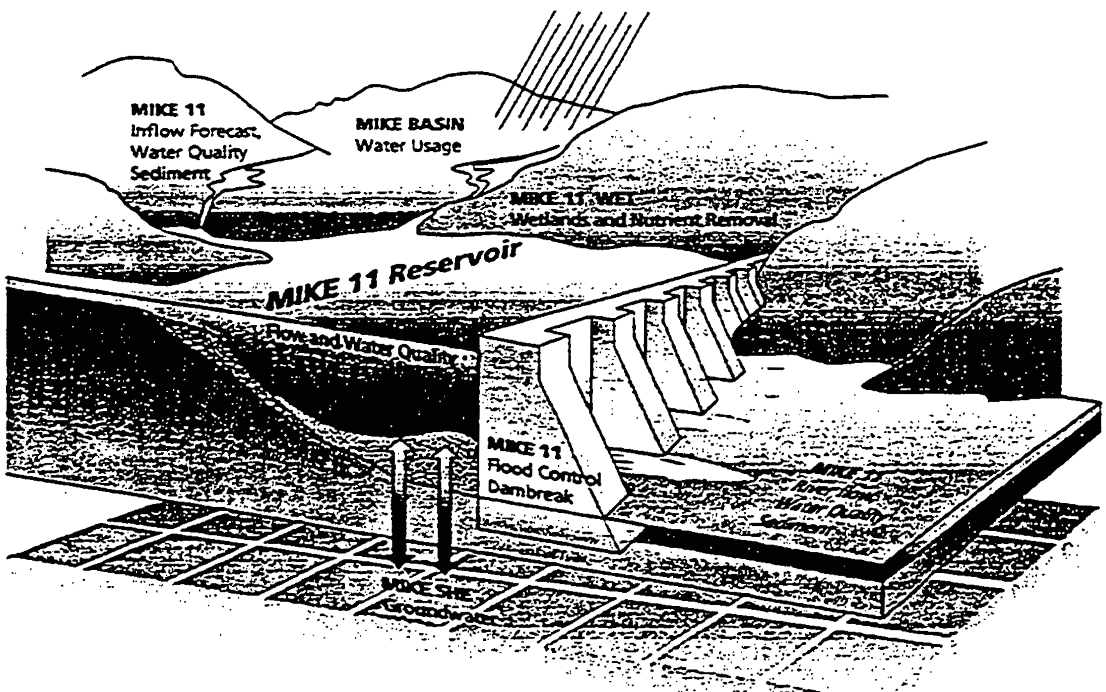


Figure 2. Model Processes.

A short description of the models is given below:

**NAM**

In a simplified quantitative form the model describes the behaviour of the land phase of the hydrological cycle /1/. Based on catchment data such as precipitation, potential evapotranspiration and temperature combined with characteristic model parameters for each storage (surface, intermediate and groundwater storage) the model computes mean daily values of runoff.

**MIKE BASIN**

The water balance model is structured as a network model in which the rivers and their main tributaries are represented by branches and nodes. The branches represent individual stream sections while the nodes represent locations, where specific water demands and activities can be specified such as eg water abstractions for irrigation, urban water supply or reservoir operations. The model concept is based on a simple water balance assumption. The simulation of a reservoir requires specification of rule curves defining the desired storage volumes, water levels and releases at any time as a function of existing storage volumes, time of the year, demand for water and possible expected inflows.

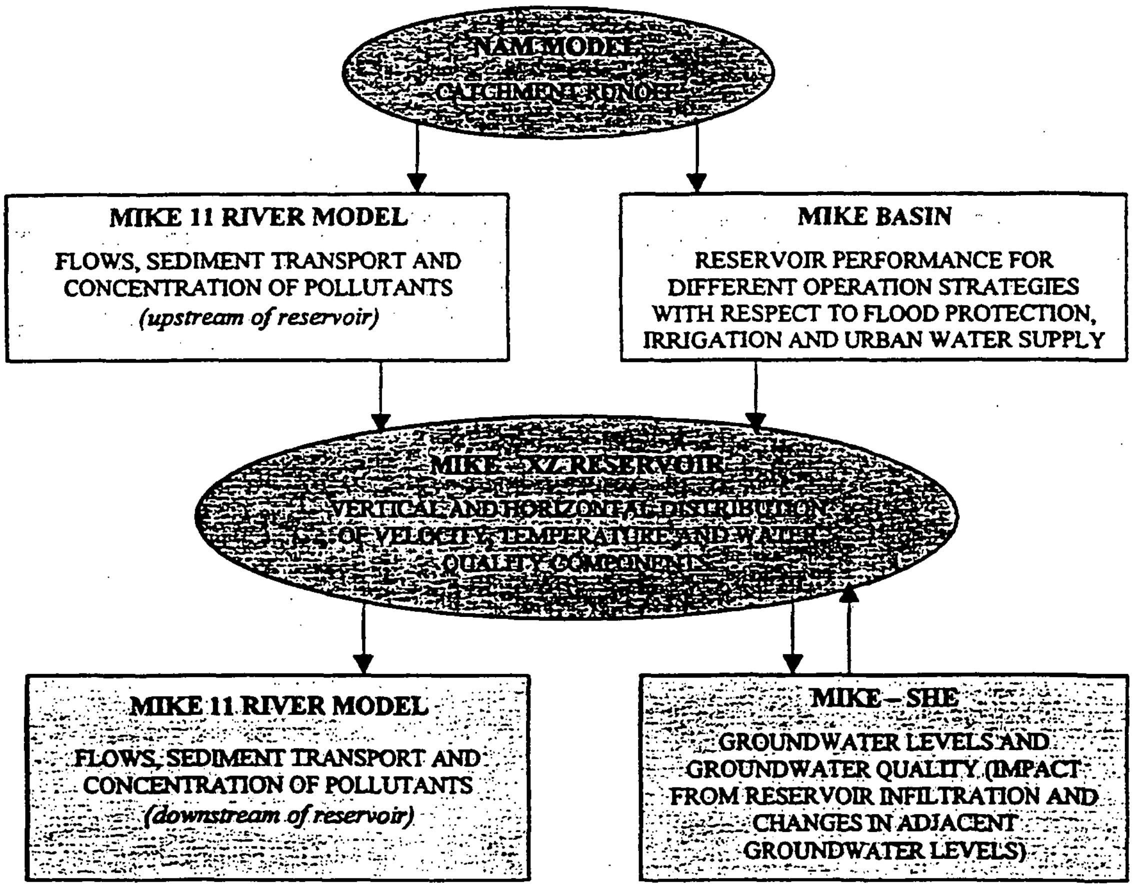


Figure 3. Integrated model structure.

**MIKE 11 River model**

The river model is a one-dimensional fully dynamic model for the simulation of flood forecasting and flood control measures /2/. In addition the model includes cohesive and non-cohesive sediment transport and water quality processes. A non-cohesive model is used for simulation of transport capacity of bed material load and morphological changes in the entire Prosna River. A non-cohesive graded sediment module is used for the simulation of river erosion downstream of the reservoir. The water quality model /3/ describes biological and chemical processes relevant for nutrient dynamics and processes affecting the dissolved oxygen conditions in the river.

**MIKE 11 Reservoir model**
A two-dimensional dynamic vertical model computing flows, transport and water quality conditions of reservoirs using a multi-layered approach /4/ utilising a computational grid adapting to variations in the free surface. Sun radiation, evaporation and convective heat transfer including a feedback loop from the water quality module providing the light extinction in the water from organic matter (plankton and detritus) is included. The simulated processes comprise development of stratification and eutrophication describing plankton, nutrient and oxygen dynamics.

**MIKE SHE Groundwater model**
A fully dynamic three-dimensional model /5/ simulating variations in hydraulic head, water flows and water storage in the entire land phase of the hydrological cycle, viz. on the ground surface, in rivers and in the unsaturated and saturated subsurface storages. Exchange of water between surface water and groundwater can take place as eg. infiltration of water on the ground surface through river-aquifer exchange. A dynamic interface between the unsaturated zone and the saturated zone is handled automatically.

## 3. Results and Discussion

The proposed dam construction is equipped with 3 overflow roller type gates and four submerged gates near the bottom of the dam. Normally flow will be released via the submerged gates each having a discharge capacity of 15 $m^3/s$ at a water level of 124 m. During floods water has to be released by lowering the roller gates. The roller gate position varies between level 121 and 124 m. Preliminary designs regarding crest dam elevations and required flood control curves are listed in Table 1.

The reservoir storage volume as function of level is shown in Figure 4.

Table 1. Dam crest elevations and flood control levels.

| *Dam Height* | *Jan* | *Feb* | *Mar* | *Apr* | *May* | *Jun* | *Jul* | *Aug* | *Sep* | *Oct* | *Nov* | *Dec* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 124.0 | 122.0 | 122.0 | 122.0 | 122.9 | 123.1 | 123.2 | 123.0 | 122.8 | 123.0 | 122.6 | 122.7 | 122.5 |
| 123.5 | 122.0 | 122.0 | 122.0 | 122.9 | 123.0 | 123.0 | 123.0 | 122.8 | 123.0 | 122.6 | 122.7 | 122.5 |
| 123.0 | 122.0 | 122.0 | 122.0 | 122.5 | 122.5 | 122.5 | 122.5 | 122.5 | 122.5 | 122.5 | 122.5 | 122.5 |
| 122.5 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 | 122.0 |
| m.o.m.s.l | m | m | m | m | m | m | m | m | m | m | m | m |

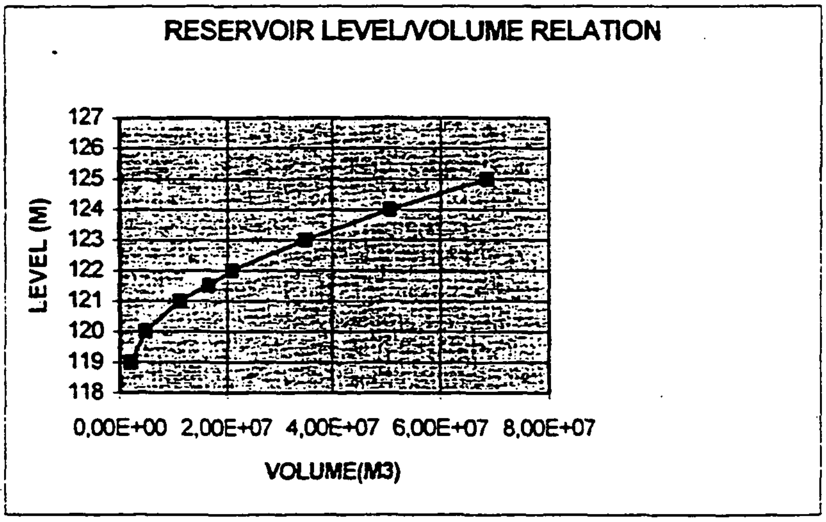


Figure 4. Reservoir storage volume as a function of water level.

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The corresponding volume of the reservoir at this level is approximately 46 million $m^3$ and the length of the reservoir is approximately 7 km. The average depth is in the order of 3 m. At the deepest point near the dam the water level does not exceed 6 m. Typical seasonal river flows in the Prosna River vary between 5 $m^3/s$ in the driest period to 60 $m^3/s$ during peak flows, as illustrated in Figure 5. The August 1985 flood with a peak flow close to 200 $m^3/s$ has been selected as design event with respect to flood control.

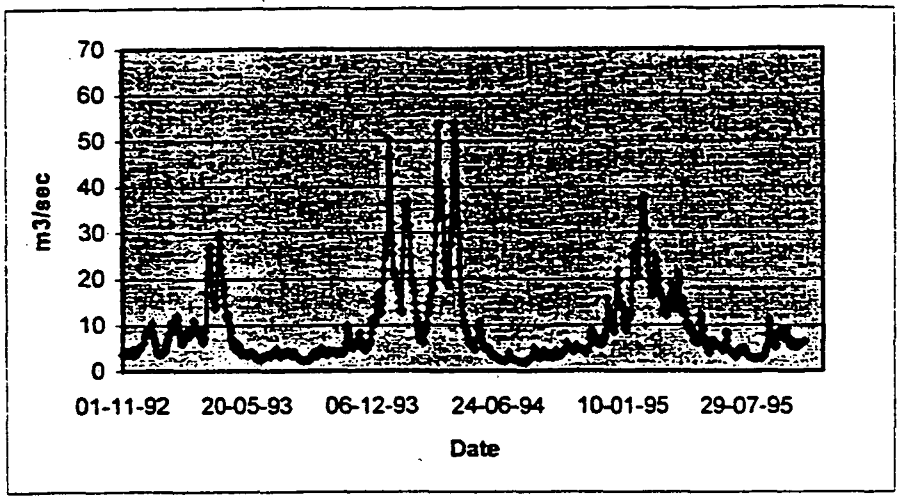


Figure 5. Recorded flow in Prosna River, 1992 –1995.

### 3.1 Water supply and flood control

Based on the preliminary design (Table 1) combined with water requirement for minimum and maximum downstream release, urban water supply and irrigation the different schemes have been analysed for monthly water balance calculations with MIKE BASIN. The simulation period covered daily flows over a period of 30 years. The calculations demonstrated that for dam crest elevations no less than 123 m the reservoir is capable of fulfilling the water demands for both water supply as well as irrigation purposes and at the same time to ensure a minimum release of water to the downstream river stretch.

Detailed modelling of a design flood (from year 1995) using the dynamic river model revealed, however, that the crest elevation should be raised to 124 m in order to avoid over topping and to ensure that the critical maximum downstream flow of 120 $m^3/s$ was not exceeded. The flood control level should not exceed 122.5 m during the expected flood period.

### 3.2 Groundwater impact

In order to investigate impacts on subsurface flow and groundwater quality in areas adjacent to the reservoir the MIKE SHE has been set-up and calibrated based on existing geological profiles and observed water table heads. In the model the reservoir is included as part of the saturated zone and the variation in reservoir operation levels (time varying head boundary) has been generated by the dynamic river model, MIKE 11. The dam has been modelled as a low permeable wall extending all the way to the level of the reservoir bottom.

Several scenarios were carried out in order to simulate impacts on groundwater levels in adjacent areas. Furthermore, scenario simulations of conservative transport of nitrate and phosphate were carried out over a long time period with a water level of 124 m in the reservoir. Spreading of contaminants from the reservoir will basically occur from the downstream end of the reservoir in the upper aquifer where large infiltration rates can move the substrate from the reservoir to the surroundings.

The simulations demonstrated that after 25 years, contamination has just started reaching the lower tertiary aquifer, though in very small concentrations (< 0.4 mg $NO_3/l$). However, due to simulated spreading of contamination below the reservoir near the banks, abstraction of drinking water in areas close to the reservoir, ie. less than 500 m from the reservoir, cannot be recommended without proper treatment. The simulations showed that the groundwater level will increase in the immediate vicinity of the reservoir particularly west of the reservoir, however, no inundated or swamped areas will arise.

### 3.3 Sedimentation aspects

The construction of a reservoir is a very significant intervention in the sediment transport regime of any river. The extremely low flow velocities in the reservoir cause the sediment transport capacity to vanish. This normally gives rise to three problems: reservoir siltation, backwater sedimentation and downstream erosion.

### 3.4 Siltation

The annual transport of wash load at Kania (ie. the sediment which will enter the reservoir) has been estimated at between 1700 and 6300 tons with a mean value of about 3500 tons. A conservative estimate of the lifetime of the Wielowies Reservoir can be obtained by assuming that all the sediment entering the reservoir will be trapped. The MIKE 11 River model has been used to investigate the extent of the trapping efficiency of the reservoir, ie. how much of the sediment entering the reservoir will deposit permanently. The simulation results show that close to 100% of the sediment will be trapped. Considering the volume of the reservoir (Figure 4) compared to the wash load it is concluded that siltation is not a problem within the lifetime of the reservoir.

### 3.5 Back water sedimentation

Assuming that the backwater sedimentation takes place over a length of say 500 m and a width of 200 m the annual sedimentation rate can be estimated at 0.15 m. This will require dredging at regular intervals. The sediment deposited in the backwater area will be sand, which will have significant value as a construction material, so the value of the dredged sediment will probably exceed the costs of dredging.

### 3.6 Downstream erosion

The reservoir will interrupt the large transport of bed material load in the river, thus the reach downstream of the reservoir will experience a severe sediment deficiency and be subjected to erosion. The erosion will depend on the composition of the material below the present riverbed. Model simulations have been carried out using the model for graded sediment. This model covers the reach from the reservoir and approximately 25 km downstream. The simulated development in the riverbed profile in the downstream reach is shown in Figure 6.

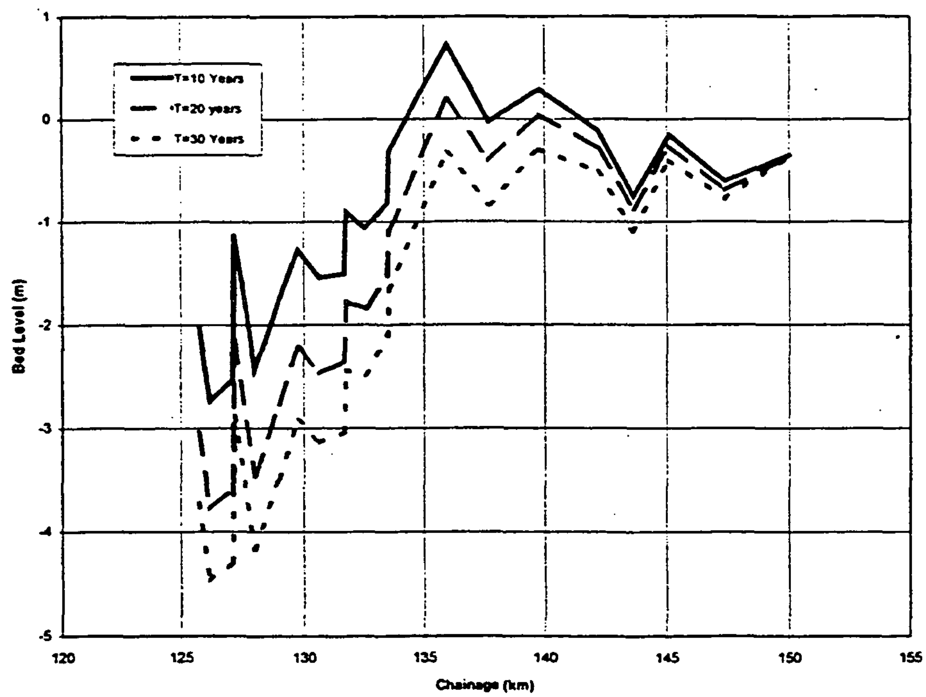


Figure 6. Simulated downstream erosion.

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### 3.7 Reservoir Water Quality

The reservoir water body undergoes biological and chemical processes, which have other characteristics than the biological/chemical processes in the river.

The reservoir modelling is based on the MIKE 11 RESERVOIR model complex. The model set-up utilises the topographical description in the MIKE 11 set-up for the Prosna River system. The topographical description of the reservoir has been imported from a DTM (Digital Terrain Model) model with a resolution of 50 by 50 m utilising an ARCVIEW interface. As the two modelling systems are fully integrated (and run simultaneously) a MIKE 11 solution is used in the upstream reaches from Mirków to the Wielowies Klasztorna Reservoir, and a full vertical solution is used within the reservoir area. The upstream river model automatically provides inflow boundary conditions for the reservoir model.

In order to simulate the exchange of air/water heat transfer, data on wind speed and direction, air temperature, relative humidity and sun radiation are given as input. The simulated hydrodynamic conditions in the reservoir form the basis for calculating the transport processes affecting the nutrient dynamics in the reservoir and the resulting algae production, chlorophyll-a levels and the oxygen concentrations. Nutrient inflow to the reservoir is simulated by the river model. In addition estimates on direct and diffuse loading to the reservoir has been included. Load estimates for the river and reservoir model were found from Polish studies /7/, /8/.

- Phytoplankton carbon (PC)
- Phytoplankton nitrogen (PN)
- Phytoplankton phosphorus (PP)
- Chlorophyll-a (CH)
- Zooplankton (ZC)
- Detritus carbon (DC)
- Detritus nitrogen (DN)
- Detritus phosphorus (DP)
- Inorganic nitrogen (IN)
- Inorganic phosphorus (IP)
- Dissolved oxygen (DO)
- Water temperature

Figure 7 shows the simulated vertical temperature distribution in the reservoir July 24th. In the upstream part, cold water enters the reservoir and tends to flow beneath the relatively warmer water at the surface. In the deeper parts of the reservoir this results in a temperature gradient of about 2-3°C between the bottom and the top waters. The simulated chlorophyll concentrations are shown in Figure 8.

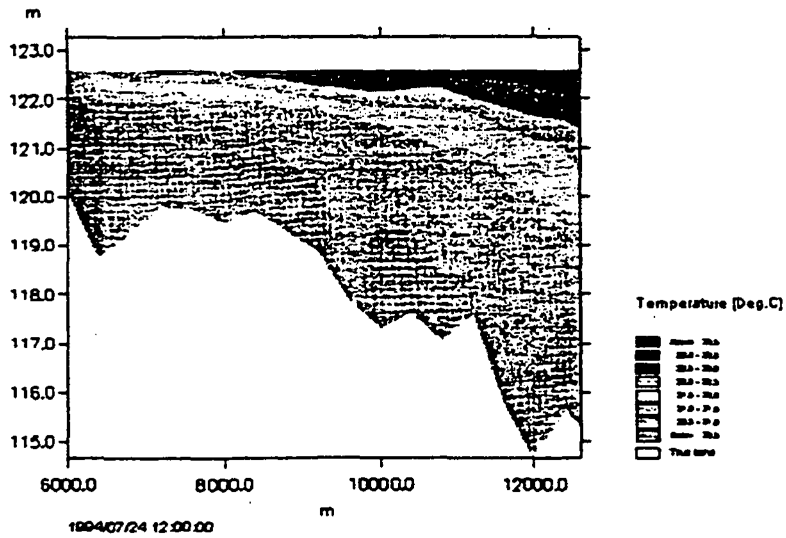


Figure 7. Vertical temperature distribution in the reservoir.

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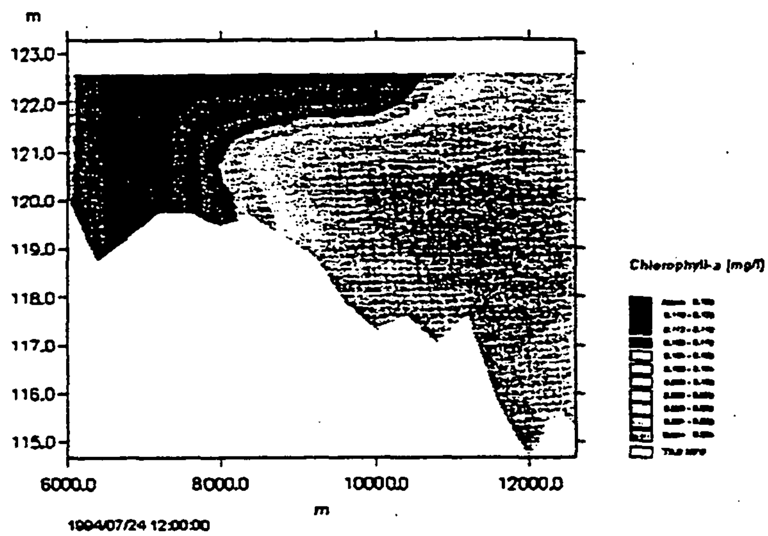


Figure 8. Vertical distribution of chlorophyll-a concentrations in the reservoir.

The vertical distribution of chlorophyll-a concentrations, shown in Figure 8, is the likely situation to occur for the Wielowies Reservoir. River water with a high concentration of nutrients enters the reservoir and as the flow velocity decreases in the reservoir, production of phytoplankton (algae) increases. The predicted levels of chlorophyll-a are in the range of 0.08-0.12 mg/l, which means that the reservoir will be a eutrophic system. By evaluating the predicted chlorophyll-a concentrations and the concentrations of dead organic matter (detritus carbon) it is concluded that the transparency in the reservoir does not exceed ½ -1 meter during the summer period.

Different scenarios each representing the identified alternatives for the reservoir size and operation have been simulated with the modelling system in order to predict and compare the water quality in the reservoir for these alternatives. In general all simulations carried out indicated that the reservoir water body will develop into a highly eutrophic lake with significant phytoplankton production during the summer period. Critical summer oxygen concentrations and periods of even complete oxygen depletion are foreseen in the bottom layers.

Different scenarios with a reduced loading of organic matter and nutrients were subsequently carried out in order to determine the appropriate reduction of the pollution sources upstream of the reservoir before obtaining an improved water quality in the reservoir. The results are shown in Figure 9 representing different load reductions.

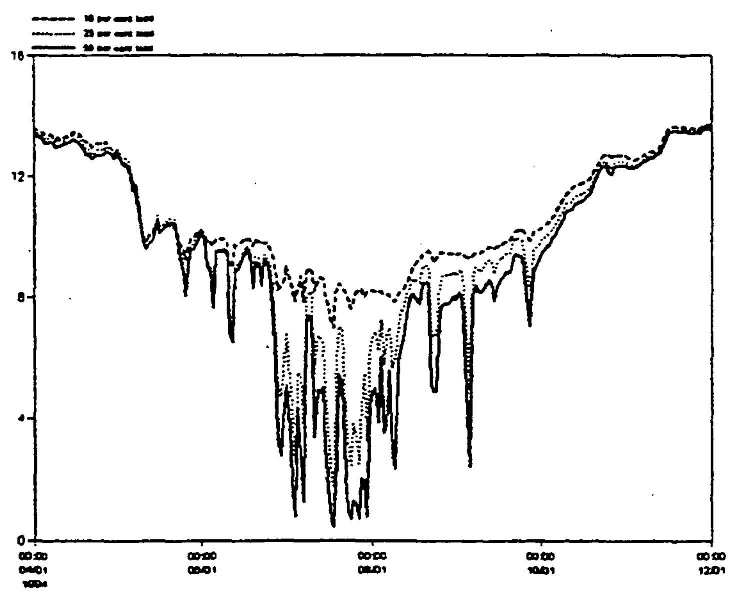


Figure 9. Time series of dissolved oxygen concentrations in the bottom water near the dam from the reduction of the pollution load: 50% (full line), 75% (dotted line) and 90% (stippled line).

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In conclusion, only the scenarios including a significant (more than 50%) reduction of the pollution load to the reservoir system generate sufficiently positive impact for the oxygen conditions. The model shows that a 50% load reduction does improve the dissolved oxygen concentrations in the bottom layers sufficiently, whereas reductions of 75% or more imply that the water quality is significantly improved, even close to the dam.

### 3.8 River Water Quality

The assessment of river water quality in the Prosna River in the pre- and post reservoir situation has been based on the dynamic river model and comprised reaches of the river influenced by the operation of the reservoir. The zone of influence comprises upstream reaches influenced by backwater effects from the reservoir and downstream reaches influenced by the quantity and quality of water released from the reservoir. The inflow from the ungauged catchment along the river has been provided via the NAM catchment model.

The released reservoir water has a negative impact on the downstream river water quality. The model results indicate that the water quality in the first 10-12 km will be negatively affected, primarily with regard to low oxygen concentrations during the summer period when bottom water is released from the reservoir. Further downstream only minor impacts on the water quality is predicted in terms of increased concentrations of ammonia and decreasing oxygen concentrations.

Simulations with different upstream load reductions to the reservoir demonstrated that the downstream impact on river water quality was limited to a short reach in the order of 10 to 15 km. Further downstream the water quality is controlled by tributary inflows and point sources

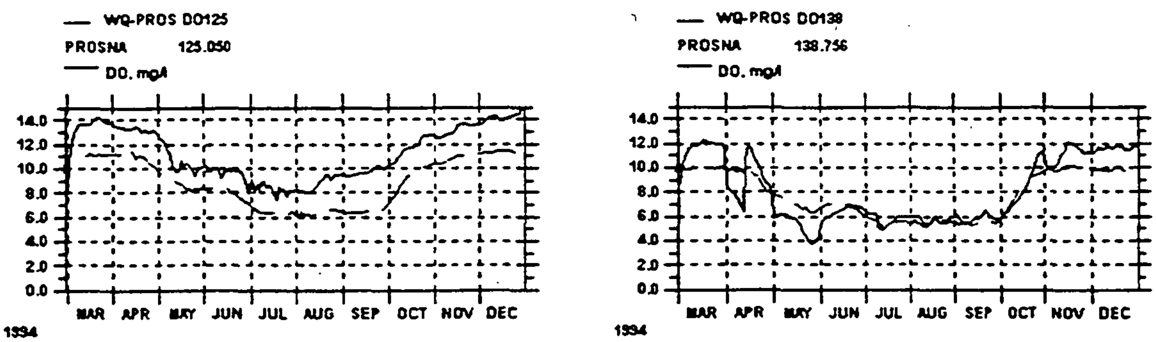


Figure 10. Time series of dissolved oxygen concentration at the dam and at Olobok. The full line represents the concentration simulated when reducing the load 90% and the dashed line represents the present situation in the river without the reservoir.

## 4. Conclusion

A new watershed modelling system has been applied for analysing the environmental impacts of a planned reservoir in Poland. The model analysis has revealed that the reservoir is capable of fulfilling the water demands of the reservoir and at the same time operates well with respect to flood control.

Only minor environmental problems with respect to groundwater are anticipated and deposition of fine sediments will not affect the lifetime of the reservoir.

The study showed, however, that the reservoir will develop into a highly eutrophic system and oxygen depletion in the bottom layers will occur regularly during the summer period. Improvements can be made only through reducing the pollution load with more than 50%.

## 5. Acknowledgement

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# Simulation of water flow on irrigation bay scale with MIKE-SHE

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# Simulation of water flow on irrigation bay scale with MIKE-SHE

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**Abstract**

This study focuses on water flow of a 9-ha experimental irrigation site in a shallow water table area in the Tragowel Plains, Australia, with the aim of quantifying the processes affecting surface drainage and groundwater levels. The irrigation site was intensively monitored to provide the required input and calibration data for modelling the physical processes. Simulation of flow processes, including infiltration, capillary rise, evapotranspiration, overland flow, groundwater exfiltration and groundwater flow into the drain, was performed using the physically-based, distributed model MIKE-SHE. The model was calibrated against the observed piezometric levels, drain flow and soil moisture data collected over a 19-month period, that represented different field conditions under seasonal changes. Model simulations were generally consistent with the observed data. However, the results highlighted inadequacies of the model, particularly in representing variations in rapid flow through macropores that occur due to the cracking and swelling properties of the soil. Despite this deficiency, the modelling study allowed quantification of the effective flow processes and provided an insight into their implications in determining the quality (i.e. salinity) and quantity of drain flow and groundwater levels. 

*Keywords:* Modelling; shallow water table; irrigation; drainage; salinity; Australia

## 1. Introduction

In Australia, deterioration of agricultural land due to salinisation and water logging has resulted in farm production losses amounting to hundreds of millions of dollars each year. Irrigated catchments in northern Victoria have been identified as some of the farming regions worst affected by high groundwater levels and salinity. In the development and implementation of management practices aimed at reducing further degradation of land and water resources, studies into the impact of farming practices such as irrigation on groundwater levels, drain flow and salinity are invaluable.

Hydrological analysis of problems associated with irrigation practices including waterlogging, drainage requirements and salinity have traditionally been carried out using simple analytical methods based on experience, e.g. Doorenbos and Pruitt (1977), as noted by Lohani et al. (1993). Recently, a few studies describing applications of hydrologic simulation models for the development of management options in irrigated areas have been reported. Nour el-Din et al. (1987) and Gates and Grismer (1989) presented hydrologic models designed to simulate the effects of management practices in salinity-affected irrigated regions with tile drainage. Nathan and Mudgway (1997) described the development of MIDASS, a

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lumped parameter conceptual model for the prediction of streamflow and salt yield from irrigated catchments with high water tables. The model was calibrated on a gauged 465-km$^2$ subcatchment of the Tragowel Plains Irrigation Area in South-eastern Australia, and was successfully applied to the Tragowel Plains to test the impacts of a number of irrigation and drainage management options on downstream river salinity.

Lohani et al. (1993) applied a comprehensive modelling approach based on the *Systeme Hydrologique Europeen*—SHE (Abbott et al., 1986; Refsgaard et al., 1992) to simulate processes at different scales ranging from a small plot (single soil profile) to a command area (hundreds of km$^2$) in an irrigated area in Central India. The study focused on variations in irrigation water requirements for various crops on the plot scale, problems of non-uniform water distribution at field scale and shortages of water supply to individual fields at command scale. Further, Singh et al. (1997) introduced a modelling system applied to a large irrigation project in Central India, that combines the hydrologic simulations of the command area and the hydraulic simulations of the canals with the use of MIKE-SHE and MIKE 11 models, respectively.

Focusing on physical processes at the irrigation bay scale, Mudgway and Nathan (1993) also used the SHE in a modelling study involving a detailed investigation on a 9-ha irrigated site with high water tables in the Tragowel Plains, Australia (Fig. 1). Using part of the experimental data collected in an intensive field monitoring program, the SHE model was used to

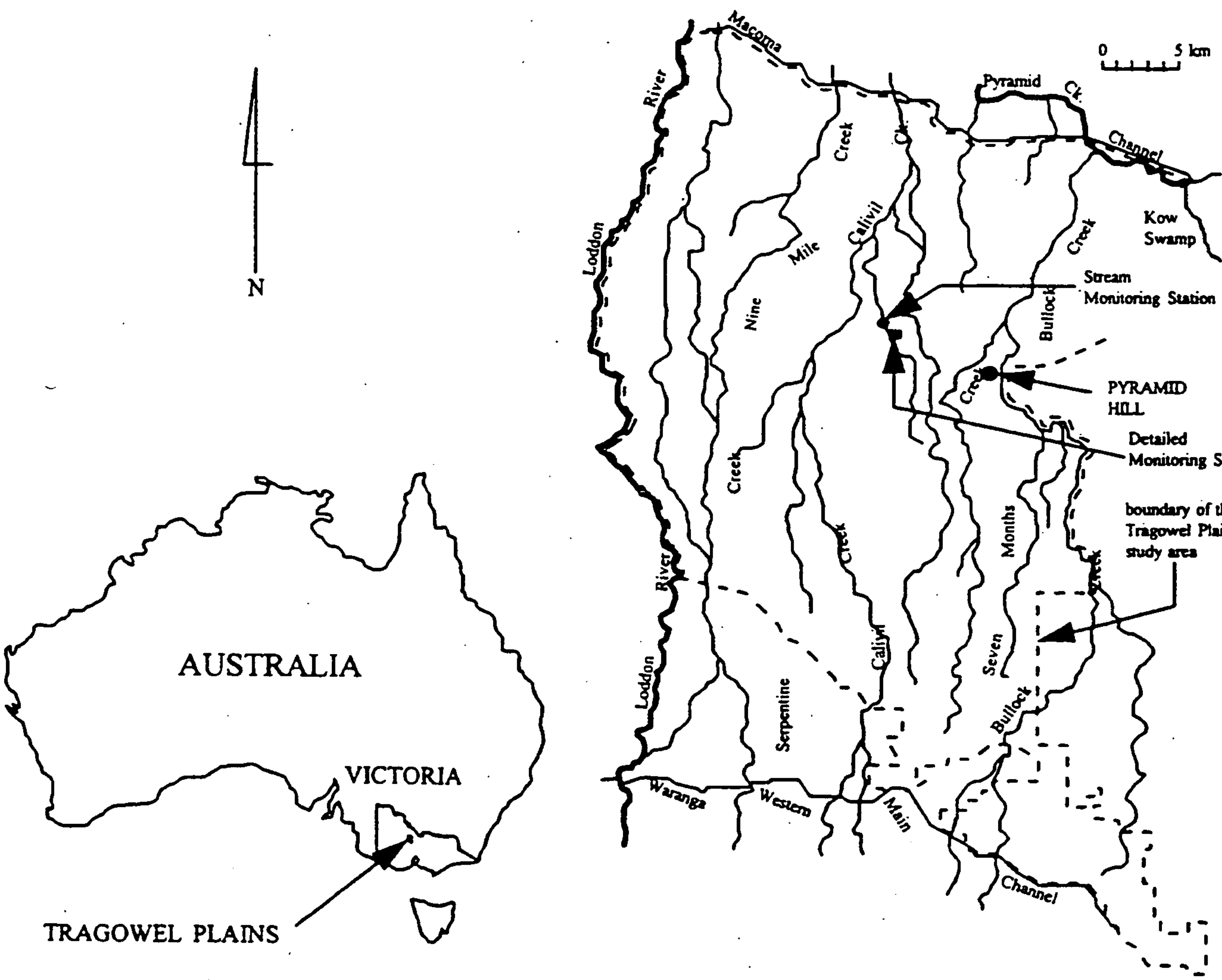


Fig. 1. Location of the study site.

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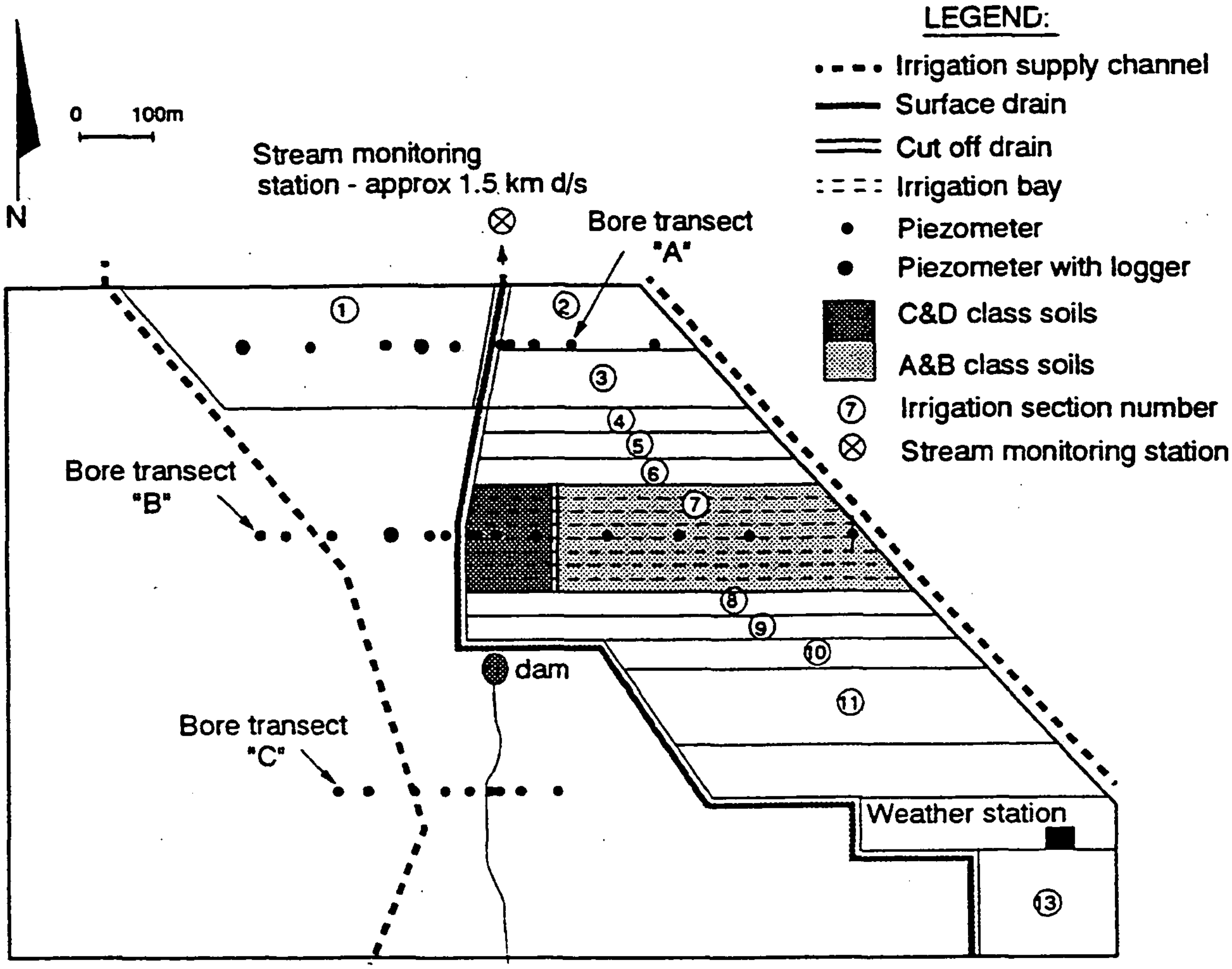


Fig. 2. Layout of the demonstration farm and locations of the irrigation site and instrumentation.

identify the significant physical processes of the experimental site and estimate the relative salt load contributions from flow components that contributed to surface drainage, as described by Mudgway et al. (1997).

This paper continues the bay-scale investigation using MIKE-SHE (DHI, 1993; Refsgaard and Storm, 1995) for the representation of the flow processes of the experimental irrigation site situated in a demonstration farm in the Tragowel Plains, Australia (Fig. 1 Fig. 2). In order to carry out a structured analysis of the hydrological processes of the irrigation bay, the present study used data collected over a 19-month period. This period represented different seasonal field conditions and the response of the study site to several rainfall and irrigation events. e study findings provide an insight into the effec- uve flow processes in flood irrigated areas and their role in transporting salts to the surface drainage, and thus contribute to an enhanced understanding required for the development of sustainable management approaches in irrigated regions.

## 2. The study site and field experiment

The 9-ha experimental irrigation site (Section number 7) considered in this study was located in a 135-ha demonstration farm, near Pyramid Hill in the Tragowel Plains of Northern Victoria, Australia (Fig. 1 Fig. 2), that was typical of salt-affected irrigated lands in the area. The irrigation site was subject to flood irrigation, and was intensively monitored to obtain time series data on piezometric levels, soil moisture, drain flows and salinity levels. It consisted of eight irrigation bays, strips of land (20 m wide and 500–600 m long) separated by check banks at right angles to the contour (Fig. 2).

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The irrigation site included all four soil classes found in the Tragowel Plains. These have been classified according to the soil salinities in the root zone as: A class (low salinity, <2400 mg/l), B class (medium salinity, 2400 to 4200 mg/l), C class (high salinity, 4200 to 5500 mg/l), and D class (extreme salinity, >5500 mg/l). In general, pasture production on the A class soils was good, and non-existent on the D class soils. About 75% of the irrigation site contained annual pasture on the A and B class soils, which was irrigated only in spring and autumn. The remainder of the site was C class soils (poor quality pasture, about 5%), and D class soils holding very little vegetation. Irrigation was applied from the eastern end of the bays via small gates on the delivery channel. The irrigation water had a salinity of approximately 300 mg/l, and shallow groundwater salinities varied, but were often high as 20 000 mg/l.

Climatic variables including rainfall, soil temperature, wet and dry bulb temperature, wind-run, and global radiation were recorded by a continuous logger at a weather station at the eastern side of the farm in which the irrigation site was located. Surface drainage from the irrigation site flows into a shallow surface drain (0.5 m deep and 1 m wide). The water level adjacent to the irrigation site, and flow and salinity at a point downstream of the irrigation site were measured continuously in this surface drain.

A shallow cut-off drain was constructed across the irrigation site west of the A and B class soils to enable runoff from the A and B class soils to be monitored separately from the highly salinised C and D soils. This is referred to as the A/B cut-off drain. The total discharge from the cut-off drain was monitored using a weir with a continuous water level gauge, and salinity was continuously monitored by a probe. Due to insufficient gradient towards the surface drain, a similar configuration could not be used to measure the runoff from the C and D class soils. Potential backflow from the drain also excluded the use of a minimal head loss device such as a flume or pipe. To overcome this, a bank was formed across the irrigation site, and the ponded water was periodically pumped into the drain after runoff events and the volume and salinity were recorded. However, due to the unavailability of a pump, on a number of occasions water volume was not measured until after several events.

Eight neutron moisture meter (NMM) access tubes were installed, two in each soil class to a depth of 1 m. Soil moisture was measured weekly at 10 cm depth increments to 60 cm and at 80 cm and 100 cm depths. Calibration curves for a similar soil at another research site were used as a basis for the calibration of NMM access tubes at this site.

The soils in the study site are predominantly heavy clays of the Shepparton Formation. The soils are developed on flood plain sediments and are subject to significant volume and bulk density changes. With the loss of soil moisture during dry periods, large desiccation cracks develop in these clayey soils, which close when the soil is wetted. These cracks can provide macro pathways for infiltration, and thus reduce the potential to leach salts as the soil matrix flow is reduced. Salts in the soils, in particular the dominant sodium salts, increase the magnitude of volume change by interacting with the charged soil particle interfaces. This is particularly important when fresh water enters a saline soil, as that produces increased ionic repulsion between clay particles, and causes swelling and dispersion of the soil particles.

The moisture characteristics and the hydraulic conductivity of the soil were determined in the laboratory using undisturbed soil cores, and the data obtained were reported by Mudgway and Nathan (1993). The soil moisture retention curves at 0.5 m depth generally showed higher soil moisture for the same tension than that at the surface or at 1.0 m. Horizontal saturated hydraulic conductivity was measured in the field with the auger hole method using the calculation described by Bouwer and Jackson (1974). The values obtained were within the range $3 \times 10^{-7}$–$7.8 \times 10^{-6}$ m/s in A/B class soils and $1.6 \times 10^{-6}$–$1.5 \times 10^{-5}$ m/s in C/D class soils.

Vertical saturated hydraulic conductivity was determined in the laboratory based on falling head and constant head permeameter methods using fresh water. After adjusting to account for the soil salinity as described by Mudgway and Nathan (1993), the values obtained were in the range $1 \times 10^{-7}$–$3 \times 10^{-7}$ m/s for A/B class soils and $3 \times 10^{-7}$–$1.1 \times 10^{-5}$ m/s for C/D class soils.

Vertical unsaturated hydraulic conductivity was determined in the field using the disc permeameter method as described in Perroux and White (1988). In order to exclude moisture from macropores larger than an equivalent diameter of 1.5 mm, a supply

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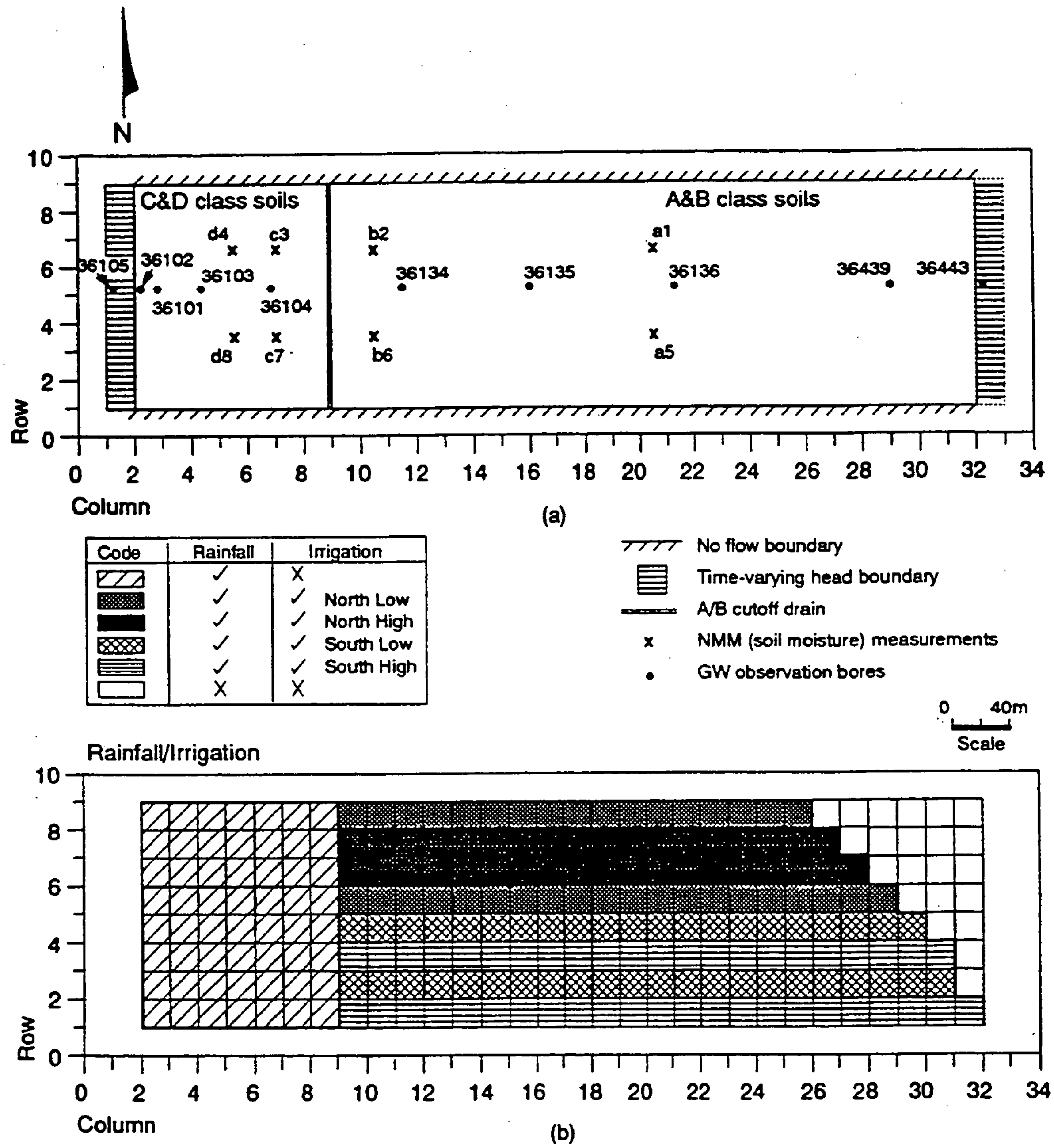


Fig. 3. Model representation of the irrigation site and locations of piezometers and neutron moisture meters.

potential of −40 mm was used (White, 1988). The hydraulic conductivity values obtained were: $8 \times 10^{-7}$–$1.5 \times 10^{-6}$ m/s for A/B class soils; and $1 \times 10^{-7}$–$1.9 \times 10^{-6}$ m/s for C/D class soils. These were higher than some of the saturated hydraulic conductivity values obtained in the laboratory, suggesting that either the core samples used in the laboratory were too small to intersect macropores, or the tension applied in the field was too small to exclude all the macropores. It can be envisaged that a significant flow component can occur via macropores while the flow through the soil matrix can be very low.

One of three transects of piezometers (3 m deep) used to monitor the water table levels over the 135-ha farm, piezometer transect B, was situated through the centre of the instrumented irrigation site (Fig. 2). Eleven piezometers in transect B were monitored weekly, while three selected piezometers were monitored with loggers at many 3-hourly intervals. Piezometer no. 36102, situated just east of the surface drain (Fig. 3), consisted of a nest of four piezometers monitoring groundwater in the upper Shepparton Formation clays at 3, 8 and 16 m, and a clayey sand aquifer at 36 m from the surface. Groundwater salinity was monitored twice per year.

In addition, leaf area index measurements were undertaken six times during the pasture growth period from February to November. A reasonably good quality data set based on the response of the experimental site to irrigation and rainfall was obtained for the 19-month period from April 1991 to November 1992.

## 3. Conceptualisation of the processes

Due to the low storage capacity available in the unsaturated zone, shallow water tables can respond rapidly to rainfall or irrigation events, causing a highly transient flow system to occur in the near-stream areas (Gillham, 1984; Jayatilaka and Gillham, 1996). In order to obtain reliable estimates of drain flow and the resulting salt contributions to waterways in areas with shallow water tables, the surface water flow and the groundwater flow components of the transient flow system need to be accurately quantified.

In general, hydrologic processes that influence the water flow and salt discharge from the irrigation site include: rainfall, irrigation, infiltration, evapotranspiration, recharge to the water table, capillary rise from the water table, groundwater flow and overland flow. Salt contributions to the drain occur through (1) the overland flow of surface runoff, (2) exfiltration—discharge of groundwater through the seepage face adjacent to the drain, and (3) base flow—direct discharge of groundwater into the drain, shown schematically in Fig. 4.

The relative significance of the processes in determining drain flow and salt loads varies according to the effective climatic factors and the geologic and topographic characteristics of the irrigation site. During summer, because of low rainfall and high evaporation, the water table is usually at its lowest level. At the supply channel end of the site, the water table can be below the height of the capillary fringe of the soil material, whereas towards the drain, the capillary

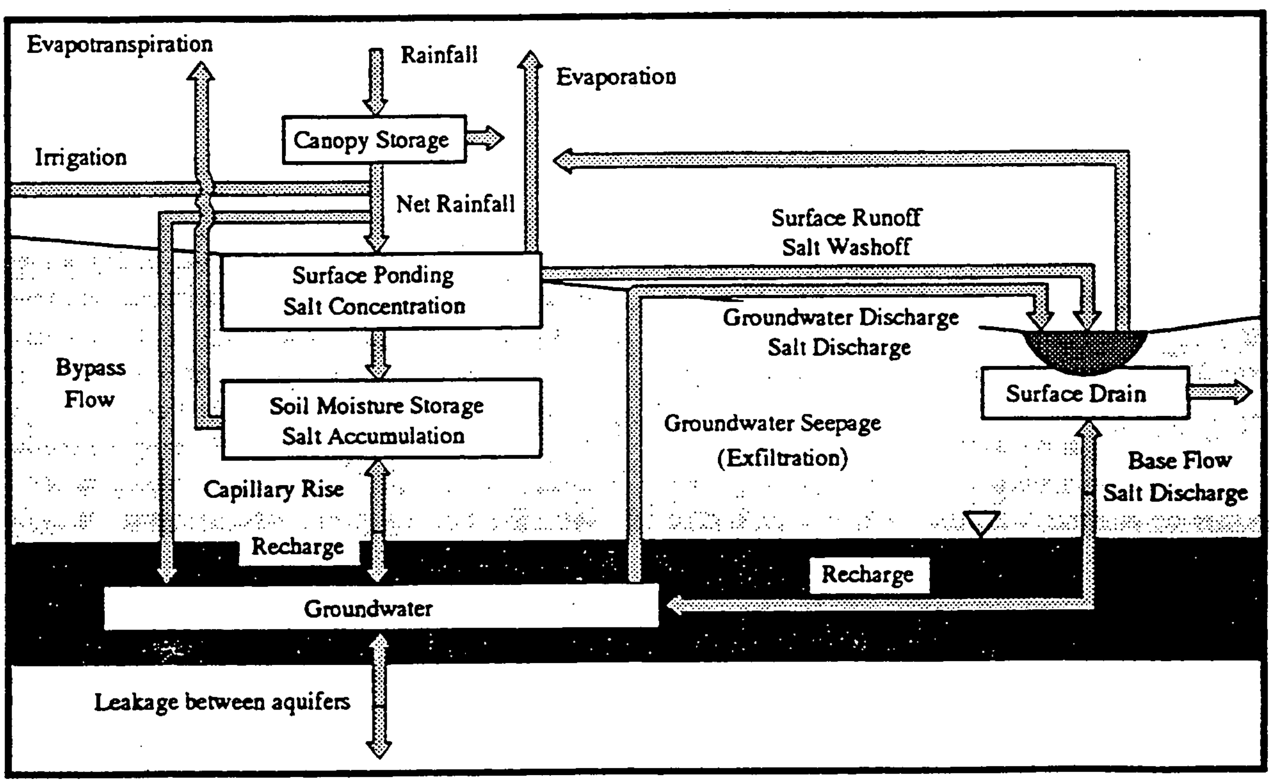


Fig. 4. Schematic of the conceptualised flow and salt discharge processes.

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·ringe extends up to the ground surface. In effect, illary up flow may continue over summer periods in the area of the irrigation site towards the drain. Capillary rise will increase the salinity in the root zone and on the soil surface as water is lost through evapotranspiration. Major summer storms may occasionally cause overland runoff, but in most cases rainfall will infiltrate. Large desiccation cracks that exist in the soil zone over summer periods allow rapid access of excess rain to the groundwater zone. Only irrigation applications or prolonged high intensity rains will cause overland flow to the drain. In the autumn, as the evaporative demand decreases, both irrigation applications and rainfall can cause significant overland flow, and such events may produce large and rapid water table rises. The rise in water table elevation will increase the potential for exfiltration in the area adjacent to the drain and direct groundwater seepage to the drain. As the groundwater is high in salt concentration, this will cause an increased discharge of salts to the drain. During the initial phase of irrigation, large quantities of salts that have accumulated at the surface during periods tween rainfall or irrigation events will be transported to the drain by overland flow. After the initial 'wash off', however, the salt concentration of surface runoff reaching the drain will be generally very low, unless the saline groundwater rises to the surface allowing transfer of salts to the surface runoff.

The hydrologic processes of the flow system that results in response to irrigation or rainfall can be considered to be superimposed on the prevailing groundwater flow system. The piezometric levels indicate that in the local flow system of the shallow aquifer, groundwater flows east to west from the supply channel end towards the drain end of the irrigation site.

During high-intensity rainfall or irrigation events, the low infiltration capacity of clayey soils can cause overland flow over the A/B class soils contributing to the A/B cut-off drain. During these periods, the soil profile near the surface can become saturated, but soil moisture depletes soon afterwards due to high evaporation demand. Very low soil moisture levels in the upper part of the soil profile are observed particu-·ly during the period from January to April (i.e. late ·ummer and early autumn). A proportion of rainfall or irrigation water is expected to reach the water table directly via the desiccation cracks without wetting the soil matrix during dry periods.

In the C/D soils, rainfall in excess of infiltration is expected to flow overland to the C/D drain. In this part of the irrigation site, the water table can rise to the surface in response to rainfall, and this will enhance saturation-excess overland flow generation. Exfiltration of groundwater through the seepage face adjacent to the drain, and the discharge of groundwater directly to the drain, form other components of the drain flow. These flow components provide important pathways for transporting salts to the drain.

## 4. Model set-up and calibration

### *4.1. The numerical model*

The integrated catchment modelling system MIKE-SHE (DHI, 1993; Refsgaard and Storm, 1995) has been adopted for this study. MIKE-SHE is a comprehensive, deterministic, physically-based, distributed modelling system for the simulation of hydrological processes of the land phase of the hydrological cycle. In the model, catchment characteristics and input data are represented in a network of grid squares, and the governing equations are solved using finite difference methods. MIKE-SHE has previously been used for studies in irrigated areas, e.g. Storm and Punthakey (1995), Singh et al. (1997).

The water flow module in MIKE-SHE is the basic module of the entire modelling system. It simulates processes of the land phase of the hydrological cycle including interception, evapotranspiration, flow in the unsaturated and saturated zones, water storage, overland and channel flow and exchange between aquifers and rivers. Descriptions of the model representation of the water flow processes of the study site are given in Mudgway et al. (1997). The model can be used to describe the water movement in the study area in general, but it has limitations particularly regarding the representation of rapid flow via macropores (e.g. desiccation cracks) in the soil zone. The model allows for an empirical representation of rapid flow through the cracks, but does not adequately represent the changing conditions in cracking and swelling soils. The equation used in MIKE-SHE allowed a fixed fraction of net rainfall or irrigation to reach the water table

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as bypass flow. The model did not allow changes in bypass flow in a manner that would be consistent with the cracking and swelling properties of the soils in the study area.

*4.2. Model conceptualisation*

Representation of the irrigation site in MIKE-SHE is shown in Fig. 3. Particularly when modelling small areas such as the irrigation site considered in this study, it is important to accurately define boundary conditions for the groundwater system. Since the local groundwater flow direction is from east to west, northern and southern boundaries that are parallel to the flow lines were assigned no-flow type boundaries. At the eastern and western boundaries, time-varying head boundary conditions were assigned. The depth of the irrigation site was taken as 16 m, and an impermeable boundary condition was assumed at the bottom of the site.

The eastern time-varying boundary (upslope end) was described by the piezometer no. 36443. This piezometer was not influenced by the irrigation applications, and therefore, it is most representative of the groundwater conditions east of the study area. To reduce the direct effect of the imposed head variations in the eastern end, the time-varying head boundary was imposed on Column 33 of the model (Fig. 3), which is located east of the farm supply channel. The area between this boundary and the farm supply channel (see also Fig. 2) was assumed to have no rainfall. Piezometer no. 36105 was assumed to represent the time-varying boundary condition at the western (downslope) end. Because groundwater levels for the piezometer no. 36443, east of the irrigation site, were not available for the early part of the modelling period, levels from the piezometer no. 36105 corrected for the mean level difference between the two piezometers (60 cm) were used to complete the data for the boundary condition at the eastern or upslope end.

The modelled area was discretised into 20 × 20 m square grid cells, each row representing one of the eight irrigation bays. The model represented the A/B cut-off drain between Columns 9 and 10, and the C/D (imaginary) drain to the west of Column 3 (Fig. 3). The community drain was not included in the model because it would have been difficult to represent both the community drain and C/D drain correctly and at the same time impose a time-varying head boundary in Column 2. The irrigation was applied as additional rainfall in all grid cells between the A/B drain and the farm supply channel according to the specified frequency and rates for individual bays (Fig. 3).

The spatial resolution of 20 m in the horizontal directions resulted in 240 internal grid cells. The vertical resolution varied with depth. In the unsaturated zone, 1 cm distance between the topmost nodes and 5 cm distance between the subsequent three nodes were used. The distance between the rest of the nodes were 10 cm. A fine nodal resolution at the surface was required to facilitate modelling of the overland flow generation. The groundwater flow was represented as a 2-D flow system in an unconfined aquifer of 16 m depth.

In the area upslope of the A/B cut-off drain, the soil was described by four soil layers with change of properties at 1, 46 and 86 cm depths. In C/D class soils, two soil layers were considered with change of properties at 35 cm depth.

*4.3. Calibration procedure*

Mudgway et al. (1997) calibrated the SHE model of the irrigation site over the period from June to September in 1991. The present study utilised the entire data set collected over the 19-month period i.e. from April 1991 to November 1992. This period included the response of the irrigation site under a number of irrigation events and different seasonal field conditions. Data available were not sufficient to undertake an independent validation process. The use of the entire data set for calibration allowed a parameter set to be calibrated that was considered applicable under changing field conditions at the study site.

Initial groundwater levels were assigned based on the piezometric levels measured on the 19 April 1991. The model parameters that describe soil characteristics were initially estimated using the measured data given in Mudgway and Nathan (1993), and were subsequently modified during calibration. Parameter values left constant during the calibration procedure included:

- detention storage of 5 mm, assumed to represent microvariations on the ground surface

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that could not be described by the 20-m model grid resolution

- roughness coefficient ($10\,m^{1/3}/s$) used in modelling overland flow, vegetation and drain characteristics, estimated from the data given in Mudgway and Nathan (1993)
- the bypass fraction that represents flow through desiccation cracks, assumed as 50% of the net rainfall or irrigation application.

A qualitative comparison of the spatial and temporal variations between the simulated variables and the observed data was used in the calibration process. Initially the calibration process was aimed at obtaining an acceptable match between the observed and simulated drain flows, as well as a good match with groundwater levels at piezometric locations.

In preliminary calibration, the observed soil moisture data at various locations within the irrigation site were not used, and the calibration was achieved in terms of groundwater levels and drain flows. Subsequently, a comparison of soil moisture data revealed a considerable disagreement, in particular during the mmer period when there were soil moisture decreases due to the high evapotranspiration demand. The reason for the apparently good match of the groundwater levels can be explained by: (1) even though the soil moisture was simulated incorrectly, the flow in the unsaturated zone was reasonably correct, particularly near the ground surface; (2) as the modelled area was small, the groundwater levels were dominated by the imposed time-varying head boundary conditions. The pattern of the recharge may be of secondary significance compared with the influence of the assumed boundary conditions. This highlights the finding that an acceptable agreement between certain measured and simulated variables does not necessarily guarantee that other variables are matching, and it is important to check the status of the other variables. The calibration was subsequently repeated, comparing soil moisture levels at different depths at several locations.

In the calibration process, qualitative comparisons between the observed and predicted drain flows appeared to be more appropriate rather than the use numerical criteria. The latter would not have provided a correct interpretation of the results because of the monitoring technique used for the C/D drain, and the problems encountered in simulating some events in the A/B drain early in the calibration period which overshadow the comparison with the other events. The flow in the A/B drain was dominated by surface flow, and therefore it was largely determined by the proportion of the rainfall or irrigation water that runs off as excess water. The most influential parameter in this respect was the vertical unsaturated hydraulic conductivity function used in MIKE-SHE that determines the infiltation capacity of the soil.

In the calibration of groundwater levels, the saturated horizontal hydraulic conductivity of the aquifer was shown to be the most important parameter. Although it should be emphasised that the boundary conditions have a pronounced impact on the simulated heads, the conductivity determines the gradient along the irrigation bays and the response in the groundwater head to temporal variations in groundwater recharge and capillary rise.

Table 1 include hydraulic conductivity values of the calibrated model. MIKE SHE uses a simple exponential function relating the unsaturated hydraulic conductivity to the soil moisture retention characteristics and the saturated vertical hydraulic conductivity, developed by Brooks and Corey (1964). The exponent ($n$) used in this function and the vertical saturated hydraulic conductivity ($K_{vsat}$) valued used in the calibrated model are given in Table 1(a). The horizontal saturated hydraulic conductivity values ($K_{hsat}$) are listed in Table 1(b).

Table 1
(a) Calibrated model parameters: vertical saturated hydraulic conductivity. (b) Calibrated model parameters: Horizontal saturated hydraulic conductivity

(a)

| Soil type | Depth (cm) | $K_{vsat}$ (m/s) | Exponent $n$ |
|---|---|---|---|
| A/B | 0–1 | $1 \times 10^{-9}$ | 10 |
| A/B | 2–46 | $4 \times 10^{-8}$ | 10 |
| A/B | 46–86 | $2 \times 10^{-8}$ | 9 |
| A/B | > 86 | $1 \times 10^{-7}$ | 7 |
| C/D | 0–35 | $2 \times 10^{-7}$ | 7 |
| C/D | > 35 | $1 \times 10^{-7}$ | 5 |

(b)

| Soil type | $K_{hsat}$ (m/s) |
|---|---|
| A/B | $9 \times 10^{-4}$ |
| C/D | $3 \times 10^{-4}$ |

It is apparent that the $K_{hsat}$ values of the calibrated model are greater than the measured data given in Section 2, whereas calibrated $K_{vsat}$ values are less than the measured values reported in Section 2. Plausible explanations for these differences include the effects of the use of a fixed bypass fraction. This allowed high macropore flow in wet conditions even when the cracks would close, requiring a greater transmissivity of the aquifer (thus greater $K_{hsat}$) to be used in the model in order to transmit accessions to groundwater. On the other hand, lower $K_{vsat}$ would indicate the requirement for reduced matrix flow, compensating for the excessive flow through macropores to some extent. Other factors that may have contributed for the above mentioned differences are discussed in Section 5 .

## 5. Results and discussion

The modelling results presented in this paper span a 19-month period. This period represented different seasonal conditions encountered at the study site and the response of the irrigation site to several rainfall and irrigation events. Fig. 5 shows the simulated and observed piezometric levels at several locations of the irrigation site. A good agreement was obtained between the observed and simulated water levels for all piezometers. Table 2 illustrates this by presenting percentages of differences between observed and simulated groundwater levels being less than given thresholds (e.g. at Piezometer b36439, 14% of the results showed a head difference less than 2 cm). Table 2 also presents the correlation coefficients for the comparisons of observed and simulated groundwater levels. The simulated groundwater heads were mostly within 10–20 cm of the observed value at all locations, and about 33% of all comparisons had a head difference within 5 cm. The simulated water levels were partly determined by the time-varying head boundaries, particularly in the areas closer to the eastern and western ends of the irrigated site. The match obtained needs to be viewed with consideration of the influence due to the boundary conditions.

The simulated results indicate that the model did not predict a correct response of the irrigation site for irrigation events in March 1992. This is attributed to the inability of the model to simulate time-varying bypass flow through desiccation cracks. A model run was carried out with an increased bypass fraction (85%), which improved the March 1992 response significantly, as illustrated in Fig. 6. The cracking and swelling conditions of the soil vary according to the soil moisture and salinity levels, and therefore the bypass fraction should also be able to vary to be consistent with the prevailing conditions.

Fig. 7 shows simulated water table profile of the irrigated site and measured water table elevations before and after an irrigation event in October 1991. The simulated results agree well with the measurements. The response of the water table to irrigation in the cells east of the A/B cutoff drain which were flooded during the irrigation event are clearly shown.

Comparisons between the measured and simulated A/B drain flow for the entire modelling period are included in Fig. 8(a), together with the observed salt load and rainfall/irrigation data. Except for the early part of the simulation period (April–July 1991), the model was able to simulate the timing and the volume

Table 2
Comparisons of observed and simulated groundwater levels

| Piezometer | Percentage of results with abs (observed head-simulated head) | | | | | Correlation coefficient |
|---|---|---|---|---|---|---|
| | < 2 cm | < 5 cm | < 10 cm | < 20 cm | < 40 cm | $R^{**}2$ |
| b36439 | 14 | 26 | 47 | 85 | 98 | 0.861 |
| b36136 | 9 | 18 | 42 | 78 | 98 | 0.803 |
| b36135 | 16 | 38 | 60 | 87 | 100 | 0.822 |
| b36134 | 17 | 32 | 61 | 87 | 99 | 0.816 |
| b36104 | 11 | 33 | 63 | 94 | 98 | 0.887 |
| b36103 | 5 | 29 | 88 | 96 | 98 | 0.924 |

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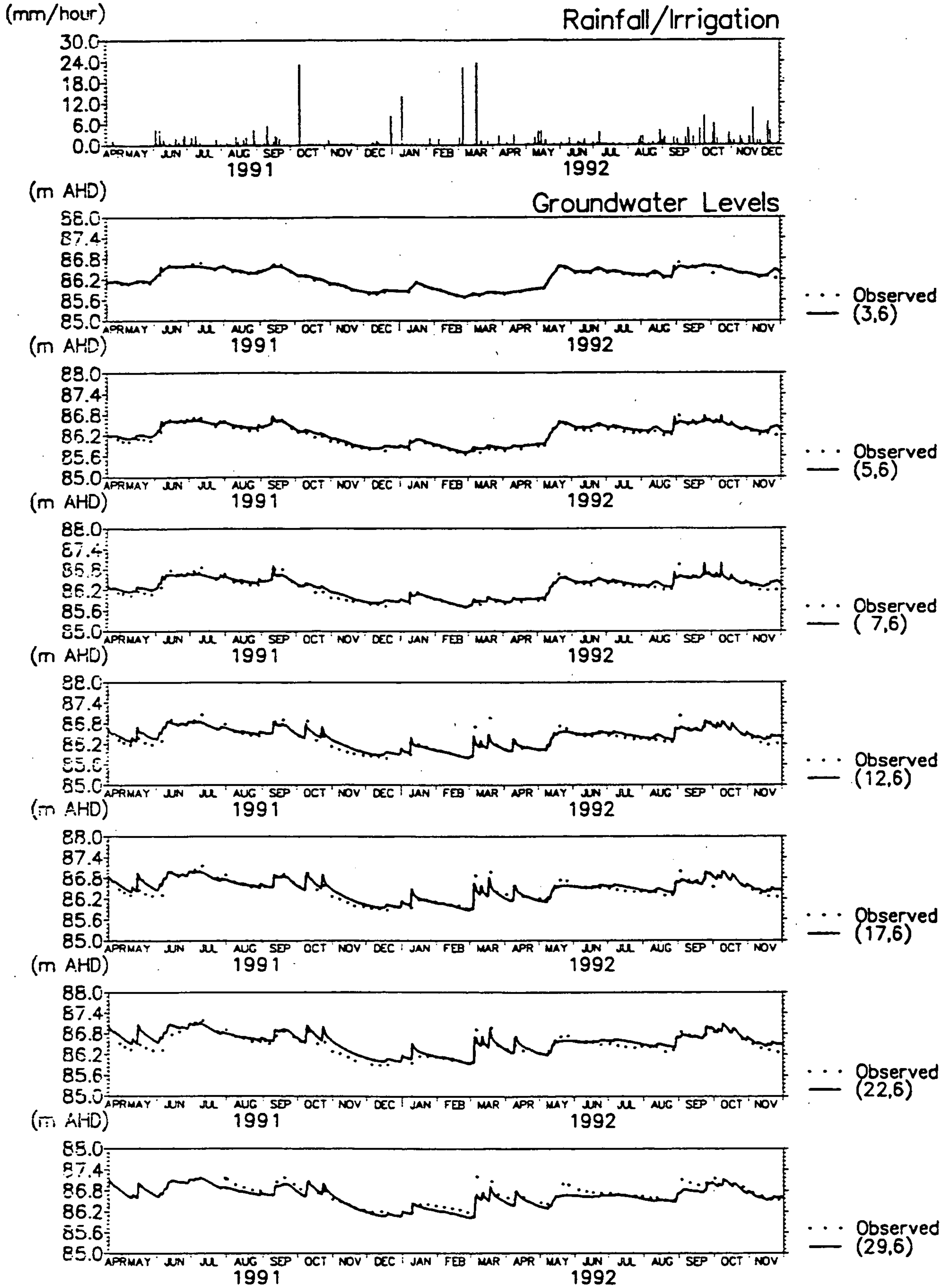


Fig. 5. Observed (dots) and simulated (line) groundwater levels at different (Column, Row) locations.



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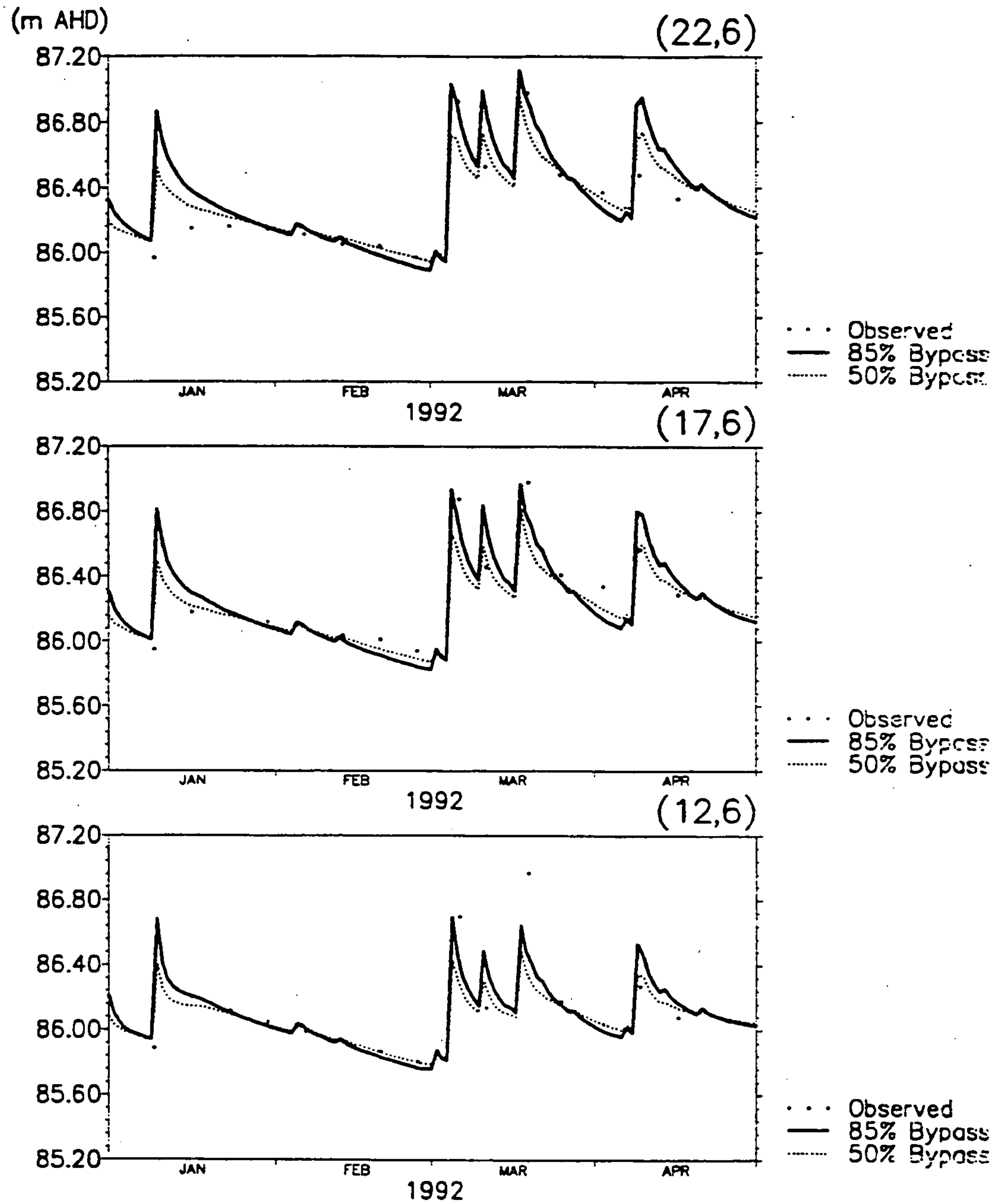


Fig. 6. Effect of the bypass fraction on simulated groundwater levels.

of flow in the A/B cut-off drain. A discrepancy resulting from the underestimation of the drain flow by the model at the beginning of the simulations is clearly visible from the accumulated A/B drain flow. A plausible explanation is that drier initial conditions were assumed by the model than existed in the field, and this led to an underestimation of drain flow during the early part of the simulation. Examination of variations in the moisture content in the unsaturated zone suggest that the discrepancy cannot be explained by dry soil moisture conditions alone, and an underestimation of the rainfall input may have contributed to the low flow estimation.

The simulated A/B drain flow, which reflects short term high-intensity irrigation events, was largely derived from overland flow. Fig. 6(a) reveals the effects of bypass flow through desiccation cracks on the runoff during March 1992, when the model over estimated the A/B drain flow. The shape of the observed flow hydrograph for the A/B drain confirms

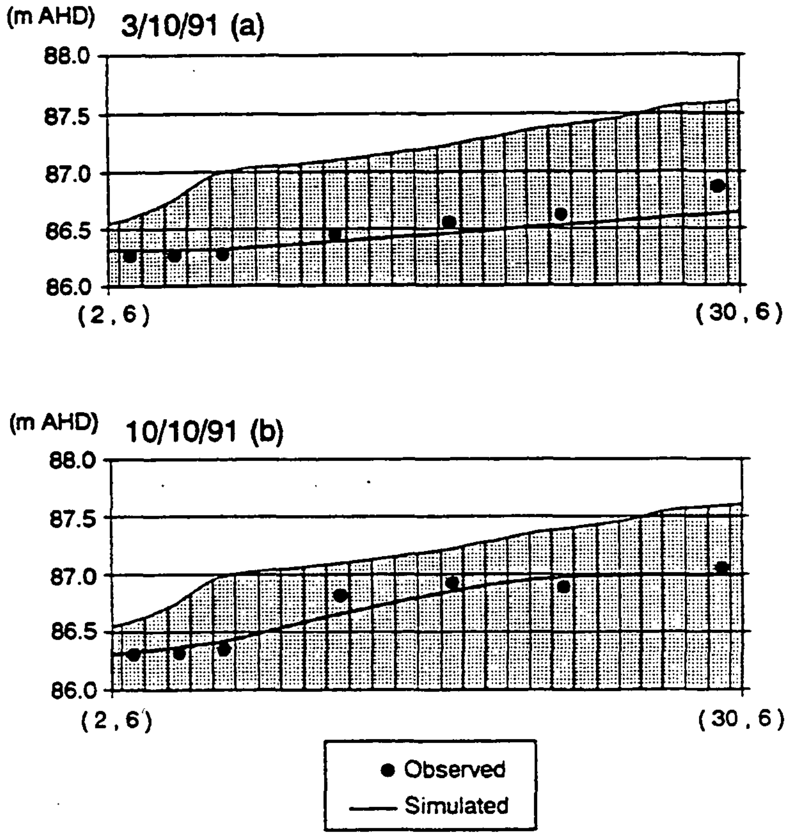


Fig. 7. Measured and simulated water table profile (a) before and (b) after an irrigation application in October 1991 (Row 6: Columns 2 to 30).

that the surface flow in this part of irrigated site was generated from the Hortonian-type overland flow and to a minor degree from the subsurface flow due to the rising groundwater. In fact, the water table rose to the level of the A/B drain only during the wet winter periods when the evapotranspiration demand was low and prolonged low-intensity rainfall infiltrated into the soil. Simulated and observed A/B drain flow from August to October in the years 1991 and 1992 are given in Fig. 8(b). The simulated flows for these two seasons agree reasonably well with the observed flows. Salt loads in the A/B drain mainly occurred during these two seasons (Fig. 8(a)), and these and the other salt load observations in April generally correlate well with some of the predicted runoff events. However, it is not clear why salt loads were not observed in the drain during the other predicted, as well as measured, runoff events that had potential for salt transport.

Fig. 9 shows comparisons of the measured and mulated C/D drain flow, observed salt load and rainfall/irrigation data. The simulated flow to the C/D drain was generated mainly from the subsurface flow caused by groundwater mounding to the east of the C/D drain. Therefore, the drain flow hydrograph of the C/D drain shows a slower responding shape compared to that of the A/B drain (Fig. 8(a)). Since no irrigation took place between the A/B and the C/D drains, very little overland flow occurred in that part of the site. It should be noted that C/D drain flow was only sampled periodically, and therefore a direct comparison between the measured and simulated results can not be made. Nevertheless, some of the predicted events correspond well with the observed flows and occurrence of salt loads in the drain.

Fig. 10 and Fig. 11 illustrate comparisons between the simulated and the observed relative saturation of the soil for seven depths at two locations, (Row 4, Column 21) within the A/B class soils and (Row 4, Column 7) in the C/D class soils. The soil moisture levels vary within the same soil class depending on the location, and the selected cells present a reasonable representation of the conditions within the soil classes. The soil moisture losses during the dry season

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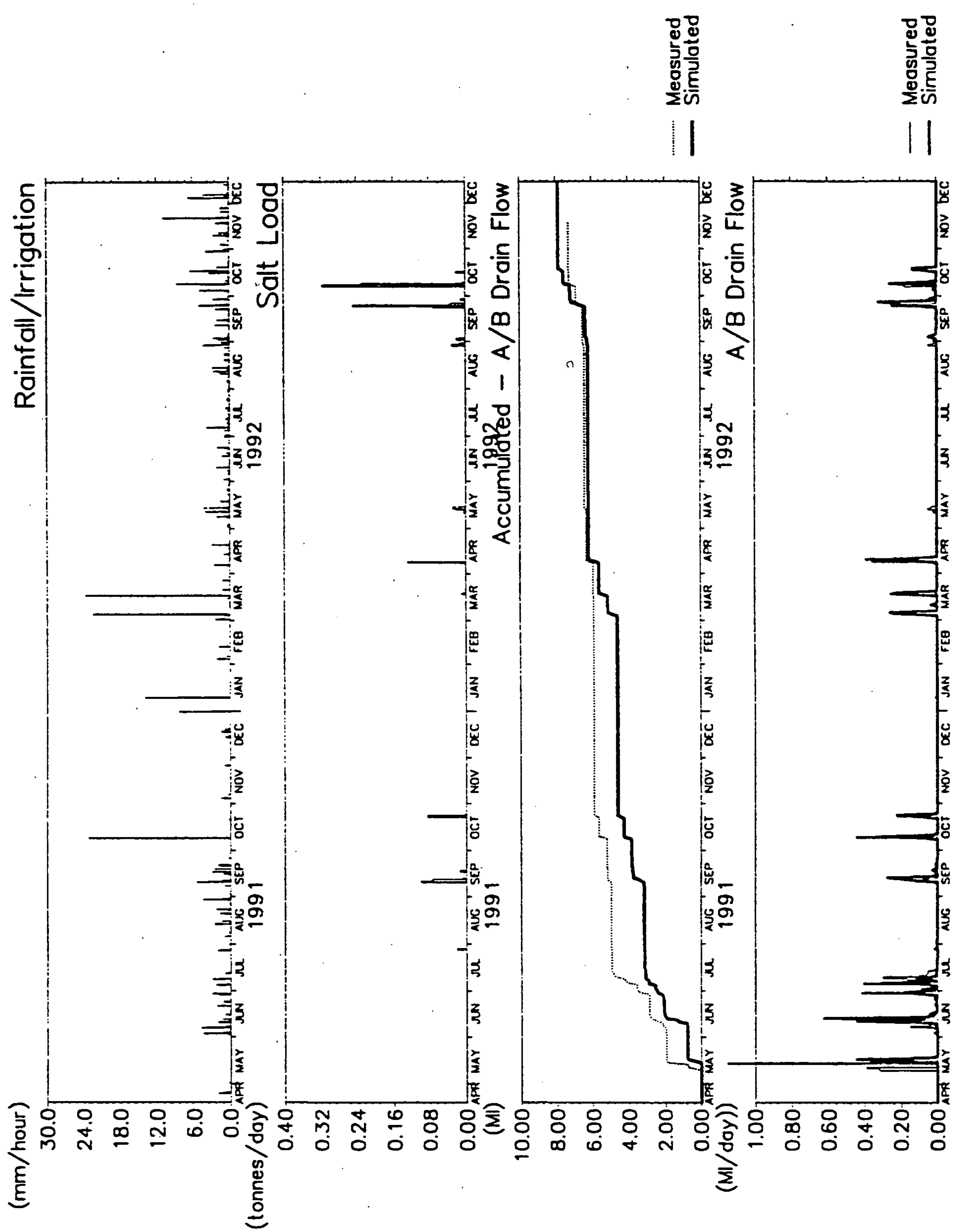


Fig. 8. (a) Measured and simulated drain flow from April 1991 to November 1992 in Drain A/B.



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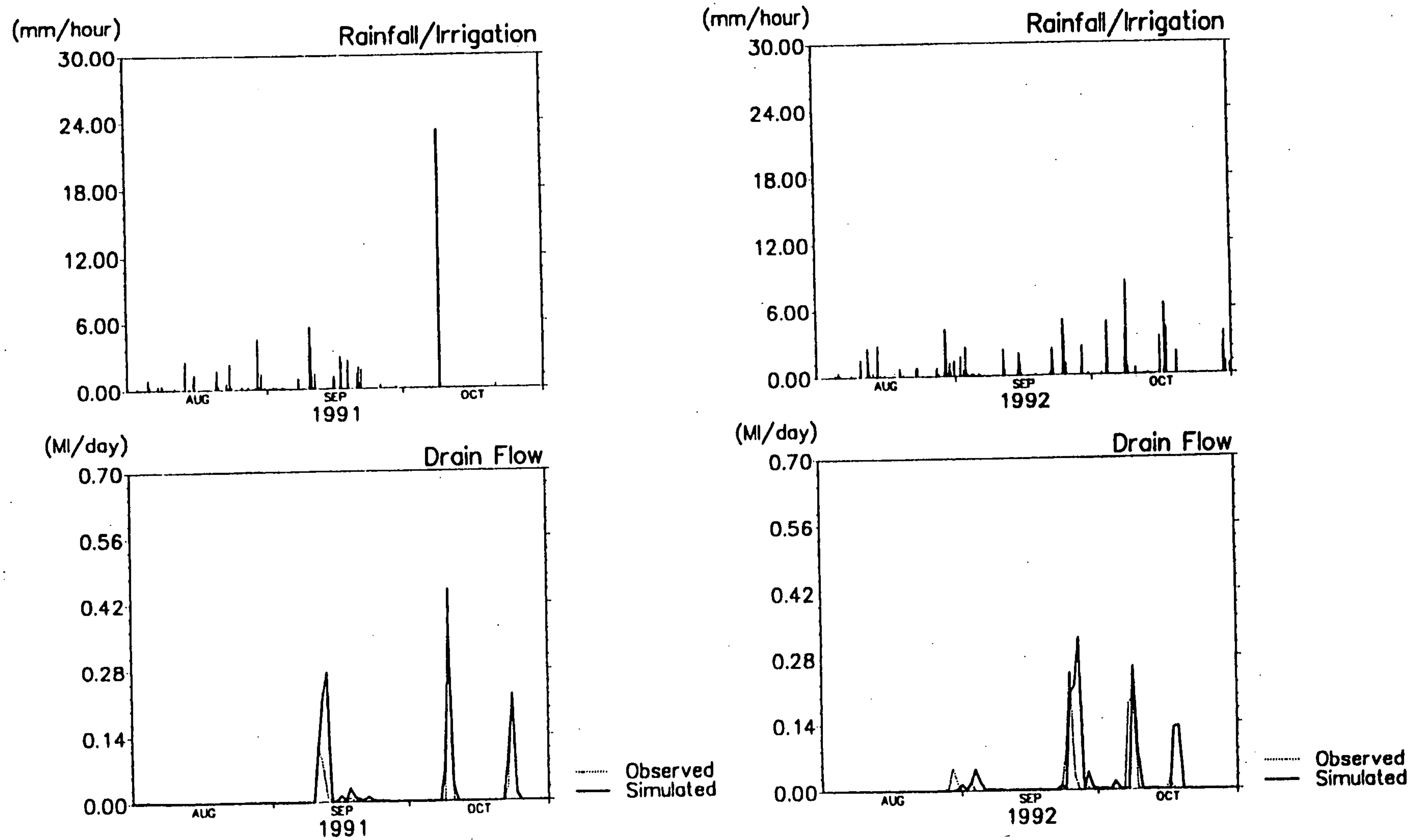


Fig. 8. (b) Measured and simulated drain flow from August to October in 1991 and 1992 in Drain A/B.

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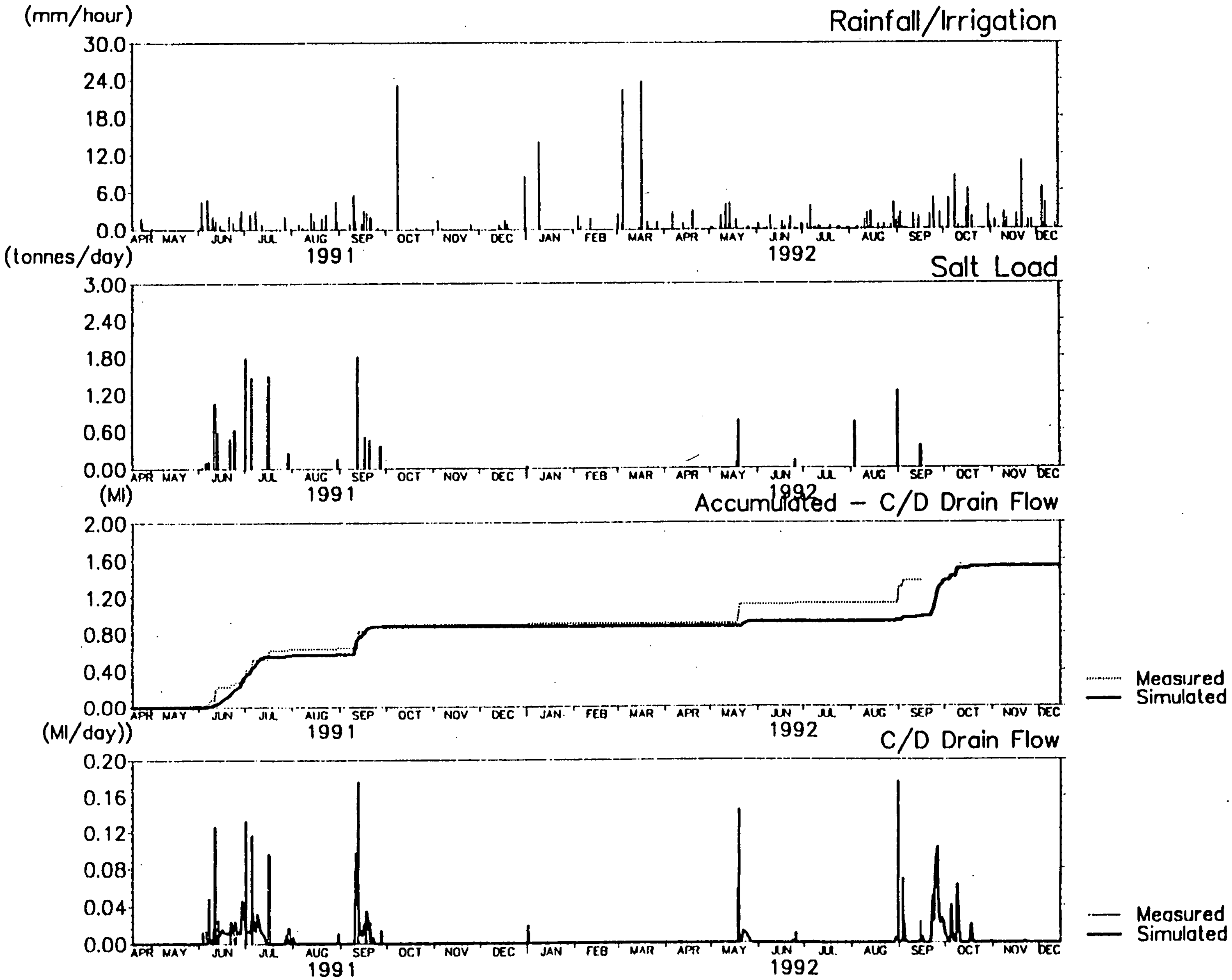


Fig. 9. Measured and simulated drain flow from April 1991 to November 1992 in Drain C/D.





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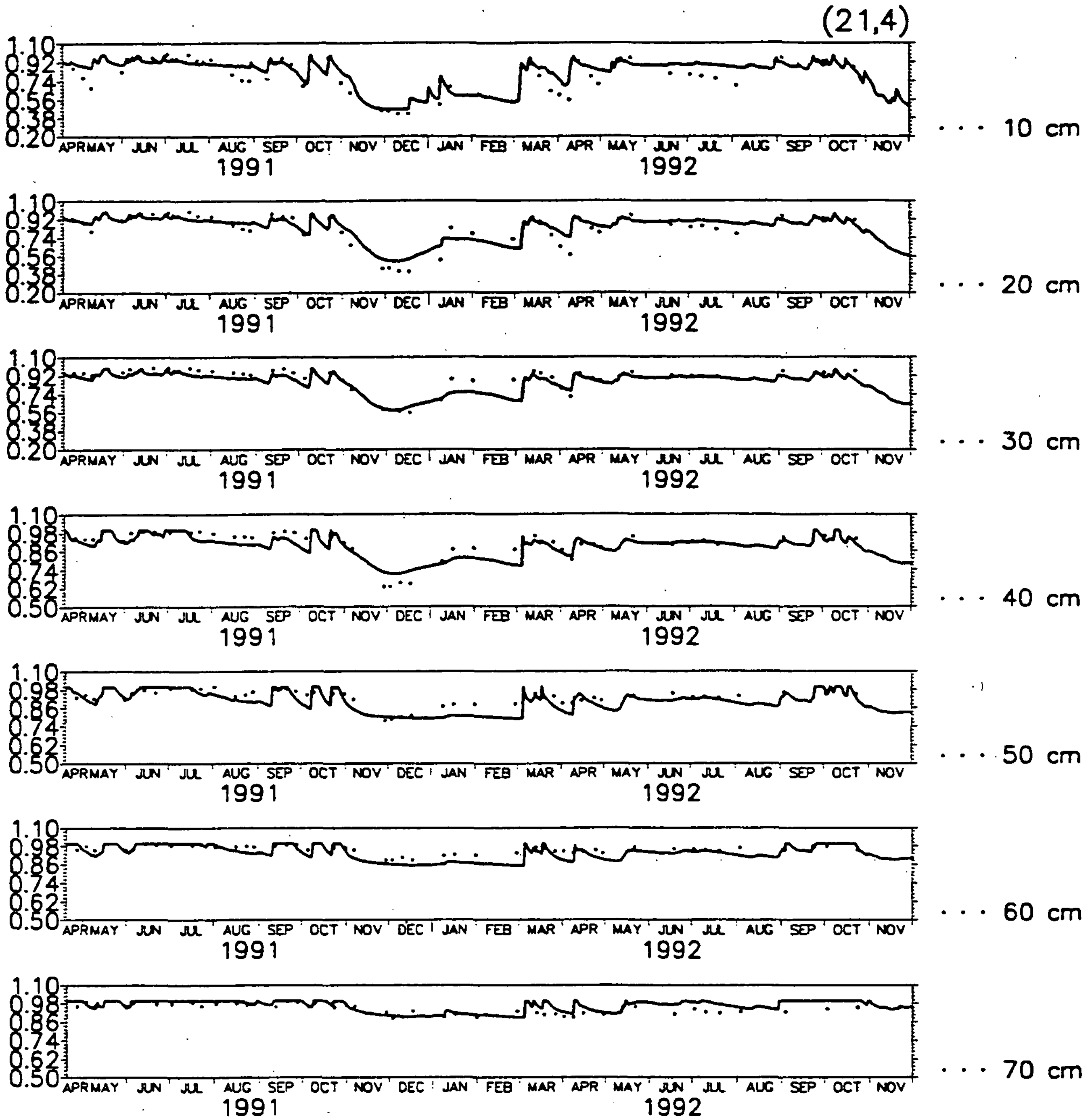


Fig. 10. Observed (dots) and simulated (line) relative saturation of soils at different depths from the surface—A/B class soil (Row 4; Column 21).

were more pronounced in the A/B class soils (Fig. 10) than in the C/D class soils (Fig. 11), particularly in the top part of the profile where a high percentage of roots were present. With increasing depth, the soil moisture loss is moderated by the influence of the capillary rise. ʾt shallow depths in A/B class soils, the large ˏcrease in soil moisture levels indicates that the capillary rise was small compared with the loss of soil moisture due to evapotranspiration. The relatively higher capillary rise in C/D class soils, which was facilitated by the shallower water table, resulted in moderating the soil moisture deficit during dry spells. The model overestimated the soil moisture loss at shallow depths of the C/D class soils. Apart from that, the model was able to simulate the relative saturation of the soils and its variations reasonably well.

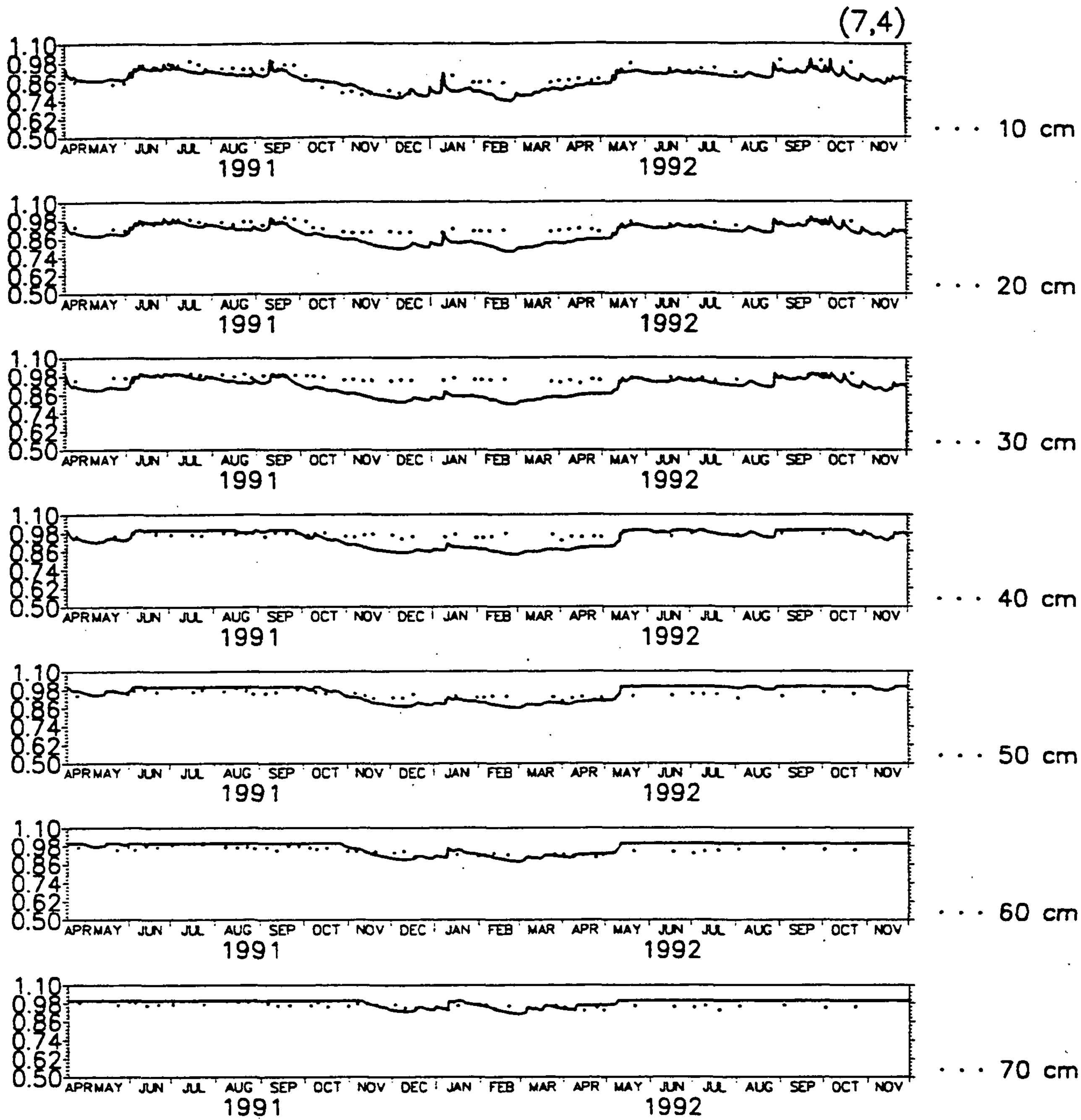


Fig. 11. Observed (dots) and simulated (line) relative saturation of soils at different depths from the surface—C/D class soil (Row 4: Column 7).

In general, the conceptual processes of the irrigation site were well represented by the model. The main difficulty encountered was the description of the soil cracking process and its influence on groundwater levels and flow processes. Fig. 12 shows the water balance for the irrigation site over the entire modelling period. It is important to note that the whole 19-month period was considered here rather than a one-year cycle. The modelling results show that inputs to the system, net groundwater inflow and rainfall and irrigation, equal the summation of outflows, evapotranspiration, subsurface flow, surface runoff, and the storage changes in the unsaturated and saturated soil zones ($dS_{uz}$, $dS_{sz}$). The groundwater flow into the irrigation site via the time-varying head boundary at the eastern end was nearly twice

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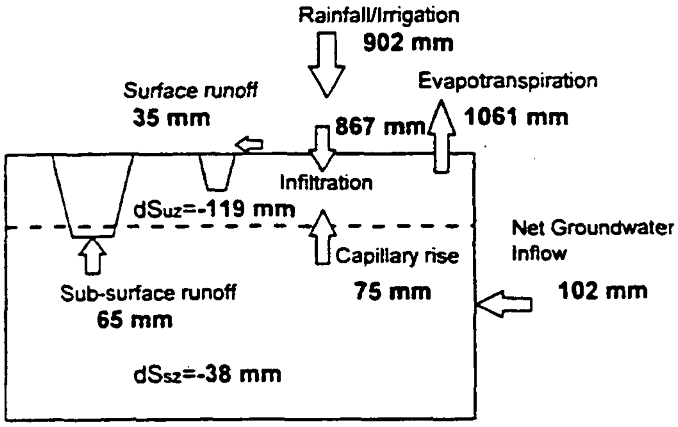


Fig. 12. Water balance for the irrigation site over the entire modelling period.

as much as the calculated net groundwater input shown in Fig. 12. This represents a significant component of the total input to the system, and highlights the influence of the boundary conditions on small modelling areas such as that considered in this study.

The results indicate that the evapotranspiration demand exceeds the rainfall/irrigation input causing a net capillary rise facilitated by the shallow water table conditions, as well as decreases in soil water storages over the simulation period. It is important to note that the water balance figures show a significant variation during the simulation period, with net recharge to the groundwater during rainfall and irrigation events and high capillary rise during dry conditions. The capillary rise over the modelling period was greater in magnitude than the subsurface flow component, and it represents an important process associated with flow systems that occur in shallow water table conditions in irrigation areas.

### 5.1. Relative flow contributions

The relative flow contributions from the individual flow processes contributing to drains are given in Table 3. The dominating flow component to the A/B drain was surface runoff (overland flow), whereas exfiltration (discharge of subsurface water through the seepage face adjacent to the drain) dominated flow to the C/D drain. The larger surface runoff contributions to the A/B drain (96%) compared with that of the C/D drain (3%) resulted from the irrigation applied over the A/B class soils. This overland flow of water can entrain the salts concentrated at the surface and provide a means to discharge salts to the drain, particularly at the start of the irrigation event. Continued salt discharge to the A/B drain via this flow component is unlikely, as the water table was at or near the surface only during limited time periods in the A/B soil area, providing less opportunity for the transfer of salt from the highly saline groundwater to the surface runoff. Although the overland runoff contributing to the C/D drain was small, as it was generated from saturation-excess runoff, continued salt discharge can occur via this flow component due to: (1) the transfer of salt from the saline groundwater that had risen to the surface, and (2) entrainment of salts brought to the surface via capillary rise in this part of the irrigation site, where the water table was at a shallower depth.

The base flow contribution to the drains depends on the head difference between the drain water level and the water table elevation, and this was small even when the groundwater had risen to the surface near the drains. In the C/D soil area, available storage space in the unsaturated zone was small and this allowed the water table to rise to the ground surface

Table 3
The relative flow contributions to drains

| Drain | Flow contributions | |
|---|---|---|
| A/B drain | Overland flow | 96% |
| | Groundwater exfiltration | 0% |
| | Base flow | 4% |
| C/D drain | Overland flow | 3% |
| | Groundwater exfiltration | 89% |
| | Baseflow | 8% |

over relatively longer time periods than in the A/B soil area. The effect of this is demonstrated by the relatively large base flow component to the C/D drain (8%) compared with that of the A/B drain (4%). Although the base flow component (direct discharge of groundwater to the drain) is small, it can cause a significant salt contribution to the drain because the groundwater salinities are often as high as 20 000 mg/l.

Groundwater exfiltration is another major avenue for the release of salts of the highly saline groundwater to the surface waterways. Exfiltration was a significant flow component (89%) contributing to the C/D drain. It is consistent with the longer time periods with saline groundwater at or near the surface in the area adjacent to the C/D drain than was the case in the area close to the A/B drain where there was no significant exfiltration.

The combined effect of the exfiltration and base flow components, which provide important pathways for salt discharge to the drain, is displayed by the larger salt loads observed in the C/D drain compared with that of the A/B drain (Fig. 8(a) and 9). The overland flow which dominated the A/B drain was not as effective as exfiltration and baseflow in transporting salts, as indicated by the relatively low salt loads observed in the A/B drain.

The results highlight the importance of quantifying the flow components of the drain flow accurately, which is an important precondition for providing reliable predictions of drain salinity levels in evaluating effects of different management options. The results also demonstrate the effects of flow processes that can become effective in flood irrigated areas with shallow water table conditions and their role in transporting salt to waterways. Exfiltration and baseflow components, which are relatively more effective in transporting salts to drains, can occur in the area adjacent to the drain. Such areas within an irrigation bay increase with decreasing depth to the water table. In addition, high water table conditions enhance the generation of saturation-excess runoff which can increase the discharge of salts to the drain. The results of this study show the importance of lowering the water tables in irrigated areas and the need for farm planning processes which can help reduce the potential for the salty groundwater to interact with the surface water systems and discharge into waterways.

### *5.2. Factors affecting simulations*

Factors that have influenced the accuracy of model simulations include the model structure and the suitability of the spatial and temporal scales to represent the dynamic physical processes, and the degree to which the model parameters were representative of the physical conditions. It became evident that the use of a constant bypass fraction of 50% in the model to represent rapid flow through macropores has considerably influenced the simulated results. This is important in the calculations of excess-runoff generation, surface ponding depths and extents, and infiltration rates. In addition, bypass flow to groundwater affects piezometric levels, and the extent of saturation in the subsurface zone, which in turn influence the exfiltration and surface runoff generation by saturation-excess. A bypass routine which can vary consistently with the cracking and swelling behaviour of the soil is required for more accurate modelling results. This is clearly evident in the model results presented (e.g. Fig. 6).

In the process of calibration, model parameters such as hydraulic conductivity values were adjusted to obtain the required results. A considerable range was evident in measured values of these parameters. This introduces an uncertainty in the validity of values assigned to a particular grid cell, the size of which is determined by the adopted scales. Some of the parameter values used in the calibrated model were outside the observed range. It is not clear to what degree the parameters used in the model represent the physical conditions.

Several factors contribute to the requirement for parameter adjustment. It can be envisaged that the model limitation of a constant bypass fraction considerably influenced the model parameters that represented soil characteristics. In addition, concerns raised by several authors regarding application of the physically-based, distributed models in field situations (e.g. Beven, 1989; Grayson et al., 1992; Grayson and Nathan, 1993) may have implications with respect to the changes required in model parameter values. For example, it has been argued that these models employ physically-based equations derived for small spatial scales, at spatial scales several orders of magnitude greater (Hughes and Sami, 1994). As it is not practically possible to estimate the model parameters the required grid element scale, field applications n enforce models to be used with a degree of empiricism. It is not clear how the equations used can describe the complex, three-dimensional, spatially heterogeneous and time varying system in the real field situations. Thus, the adjustments needed for model parameters to obtain the presented results may, at least in part, reflect the effects of these uncertainties.

Other factors with influence on the accuracy of simulations and the applicability of the results to field conditions include: leakage from the supply channel, influence of the community drain, modelling irrigation applications and the experimental set-up of the irrigation bay. Leakage from the supply channel or from the drain was not represented in the model as it is not significant in this field situation. The supply channel was not lined; however, clayey soils in this field situation and siltation over long periods of time would not allow any significant flow through the channel bed or banks. The community drain only flows during and just after irrigation events, and would not have significant impact on the water table elevation within the rigation bay. The irrigation applications are concep-'ly simulated as additional rainfall input over the ..dre irrigation bay. Although this may not represent in detail the field situation during flood irrigation, the model treatment of irrigation applications is not considered to introduce significant error as the irrigation water would reach the drain end of the bay within a reasonably short time after the initiation of the event.

A standard irrigation bay set-up is different from the experimental irrigation site considered in this study in which the irrigation excess was collected by the (A/B) cut-off drain. Without the cut-off drain, the excess runoff from irrigation would flow over the C/D soils, allowing prolonged surface ponding in this area. In effect, generation of saturation-excess runoff would be enhanced providing a greater opportunity for the transfer of salts from the saline groundwater to surface runoff. This would increase the salt discharge to the drain, as confirmed by Mudgway et al. (1997) using simulations with the cut-off drain removed.

## Summary and conclusions

The MIKE-SHE modelling study described in this paper attempted to quantify the effective flow processes contributing to surface drains of the 9-ha experimental set-up in the Tragowel Plains, Australia. The model calibration involved experimental data collected over a 19-month period. The data set included information on the response of the irrigation site to several rainfall and irrigation events under different seasonal conditions. The model parameter set derived through the calibration process was considered to be applicable under the varying field conditions at the study site. The model output provided spatial and temporal variations of variables including: water table elevation, drain flows and soil moisture levels.

The calibration was initially performed by comparing groundwater levels and drain flows. The model output was found to agree with the observed piezometric levels and drain flow measurements. However, the simulated soil moisture levels did not agree with the field data. The calibration process manifested that an agreement between certain type(s) of simulated variables and observed data does not necessarily guarantee a good match between other types of variables and observed data. The calibration was subsequently completed including soil moisture comparisons.

In general, modelling results were in reasonable agreement with the observed data, providing evidence on the ability of MIKE-SHE to handle the conceptualised processes of the physical system. The results also illustrated the need for the model to better represent the transient effects of the cracking and swelling properties of the soils in the study area.

The predicted groundwater levels agreed well with the observed data. The influence of the time-varying head boundaries at both ends of the site on the modelled water levels was identified as an important factor that needs to be considered when analysing the modelled results. The simulated relative saturation of the soil was in reasonable agreement with the field measurements. The effects of the roots of pasture, and the shallow depth to the water table were evident when comparing the wetting and drying of the soil profile in the two different soil classes.

The water balance calculations over the entire modelling period indicated the relative importance of different flow processes. The impact of the time-varying head boundary condition which allowed significant inflow into the irrigation site, and the

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importance of the capillary rise in shallow water table conditions were evident from the results.

The model was able to reproduce drain flows in reasonable agreement with the observed data. However, some discrepancy was apparent between the measured and simulated flows, particularly with respect to the A/B drain in the early part of the simulation period. The modelling results showed that flow to the A/B cut-off drain was mainly generated by the Hortonian-type overland flow resulting from short-term high-intensity rainfall or irrigation events. By contrast, C/D drain flow was mainly generated from the exfiltration of subsurface water. Groundwater discharge through the base flow component to both drains were small due to the small head difference between the drain water level and the water table.

Salt discharge to the A/B cut-off drain mainly occurs due to the washoff of salt concentrated on the surface by the overland flow. This is considered to be small compared to the transport of salt to the C/D drain via the exfiltration of saline groundwater and base flow that allows direct discharge of groundwater into the drain. The observed salt loads in the drains confirmed this, and highlighted the importance in accurately quantifying the flow components contributing to drains as a precondition for reliable prediction of drain flow salt loads under different management scenarios.

It was noted that in a standard irrigation bay without a cut-off drain, enhanced generation of saturation-excess runoff would increase the discharge of salts to the drain, due to the increased opportunity for the transfer of salts from saline groundwater to surface runoff.

This study demonstrated the effects of flow processes that can occur in flood irrigated areas with shallow water table conditions and their role in transporting salt to waterways. The study results showed that the exfiltration and baseflow components, which are relatively more effective in transporting salts to drains, can occur within the irrigation bay in the area adjacent to the drain. Shallower water table conditions can increase the area of the bay in which these processes can become effective. In addition, such conditions would enhance generation of saturation-excess runoff which can increase the discharge of salts to the drain. This highlights the importance of lowering water tables in irrigated areas and the need for the development of practices which can reduce the potential for the salty groundwater to interact with surface waters and discharge into waterways. It is important that the areas of irrigation bays with very shallow water tables are minimised through the adoption of appropriate farm planning processes.

Several factors associated with MIKE-SHE, and also generally with the physically-based, distributed models, have implications on the accuracy of the simulated results. These include: inadequacy of the model to represent the complex behaviour of the physical system, suitability of the adopted spatial and temporal scales to represent the dynamic flow processes, and the degree to which the model parameter values are representative of the physical conditions. Despite these uncertainties, this modelling study has provided an insight into the effective flow processes of the irrigation site and their implications in determining the quality (i.e. salinity) and quantity of drain flow.

## Acknowledgements

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# Evaluation of a Stepwise Procedure for Comparative Validation of Pesticide Leaching Models

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## ABSTRACT

**Four pesticide leaching model codes (PELMO, PESTLA, MACRO, and MIKE SHE) were evaluated and compared through a rigorous validation procedure combined with the application of statistical evaluation criteria. The validation procedure followed a strict stepwise approach based on suggestions put forward by a European work group on regulatory use of pesticide models (FOCUS). The experimental background comprised two different types of data sets. A laboratory and field lysimeter experiment were conducted on a Danish macroporous sandy loam soil. The aim of the study was to evaluate whether a priori model calibration on controlled laboratory data could improve the physical description of the flow and solute transport in the soil and hence the performance of uncalibrated models for predictions of field lysimeter data. The validation procedure proved to be valuable in terms of ensuring process-based evaluation of model performances and consistent model comparisons. Controlled laboratory experiments and lysimeter experiments consistently showed very significant influence of preferential flow on water and solute transport. Model codes including a description of preferential flow processes (MACRO and MIKE SHE) required less calibration efforts to meet the selected performance criteria on the investigated soil type than those without such description (PELMO and PESTLA).**

Recent findings of pesticides in drinking water wells, streams, and shallow groundwater worldwide (Fielding, 1991; U.S. Environmental Protection Agency, 1990, 1992; Legrand et al., 1992; GEUS, 1995) have focused on the methods and assumptions used for pesticide registration. The registration procedure addressing the risk of groundwater contamination employed by national authorities is often based on the determination of pesticide mobility in soils using adsorption and degradation characteristics derived from laboratory and lysimeter/field tests. If the active ingredient (a.i.) following a laboratory test leaches in quantities exceeding a certain amount, supplementary evaluations must be conducted to assess the mobility under more natural conditions through lysimeter or field tests. Such tests are expensive and time consuming and only describe a limited range of climate, soil type, and crop conditions. A way of taking this aspect into account in the registration procedure would be to use dynamic numerical models describing pesticide leaching. The major benefit of models is their potential ability to take into account the variation in time and space under a more diverse range of conditions than can usually be produced in laboratory or field experiments.

At present only few countries use simulation models in registration procedures (i.e., the USA, the Netherlands, and Germany). However, European Union (EU), legislative initiatives will oblige member states to use *suitable* calculation models *validated* at Community level in the future (EC directive 91/414, Annex VI 94/43). At present no such validated models exist and the directive itself provides no guidelines for appropriate model selection, validation, or application. As a result, several workgroups under the frame of the FOrum for the Co-ordination of pesticide fate models and their Use

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**Abbreviations:** EU, European Union; LUC, large undisturbed columns; HPLC, high pressure chromatography; TDR, time domain reflectometry; RMSE, root mean square error.

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(FOCUS) was formed to provide guidance to member states, the European Commission and Industry on the appropriate role of modeling in the EU registration process. In the final report of the workgroup focusing on leaching models special concern is addressed to the validation issue and in particular to the necessity of using a rigorous validation procedure (FOCUS, 1995).

Several pesticide leaching model codes exist at different levels of complexity (Calvet, 1995; FOCUS, 1995), each having a claimed/theoretical range of validity, defined as "the part of reality to which the validation of a model applies" (Loage and Green, 1991; FOCUS, 1995). This implies that a model code cannot be generally validated but must be tested under all the conditions for which it will be used. In practice, a given model code used for registration purposes will have to simulate several different soil-climate-crop combinations (site-specific models) all of which in principle must be validated separately. Several studies aiming to test or validate model performance under various conditions are reported in the literature (Pennel et al., 1990; Sauer et al., 1990; Boesten and van der Linden, 1991; Boekhold et al., 1993; Bergström and Jarvis, 1994; Bosch and Boesten, 1994; Walker et al., 1995; Brown et al., 1996; Groen, 1997). However, such studies are difficult to compare and evaluate as the experimental conditions, parameter availability, and validation methodology are often different, and the experimental background may not originally have been designed with model testing/validation as the main objective. A study attempting to determine the validation status of four pesticide leaching model codes for conditions prevailing in Dutch agriculture and horticulture concluded that the validation status was generally unsatisfactory (Bosch and Boesten, 1995).

In general, two types of experiments are used for testing and validation of pesticide leaching models, both having strong and weak points. Field studies with evaluation of predicted pesticide concentrations in the soil profile as used in Pennel et al. (1990), Parrish et al. (1992) and Walker et al. (1995), represent a situation close to natural conditions and provide the opportunity to evaluate the influence of spatial variation in soil physical and chemical properties on pesticide behavior in the soil (Flury, 1996). This type of study also allows rigorous evaluation of the predicted mass, position, and dispersion of a solute pulse moving through the soil profile but has limited value for evaluation of predicted pesticide leaching, as the overall mass balance of the pesticide is difficult to measure in field experiments. Laboratory/lysimeter studies as described in Sauer et al. (1990), Bergström and Jarvis (1994), Jørgensen et al. (1998b) on the other hand focus on evaluating water and solute fluxes under more controlled conditions representing a less natural situation with artificial boundary conditions. These studies are less costly than field studies and seem more appropriate for registration purposes where evaluation of the risk of groundwater contamination is the main issue. The main question, however is whether laboratory/lysimeter experiments are representative of field conditions, and whether an ideal model validation should also address this issue (FOCUS, 1995). Only few modeling studies, (e.g., Sauer et al., 1990), include both types of experiments for evaluation of the prediction of both mobility and distribution of pesticides in the soil profile.

Comparative studies of model codes using the same experimental conditions give a more precise picture of differences in model capabilities. Examples of such studies are Pennel et al. (1990), Bergström and Jarvis (1994) and Walker et al. (1995) of which only the work presented in Bergström and Jarvis (1994) had evaluation of model applicability for regulatory purposes as the primary objective. Of these studies the validation methodology is only reported in Bergström and Jarvis (1994) which include both blind and calibrated simulations in the test, whereas statistical evaluation of model performance is applied in Pennel et al. (1990) and Walker et al. (1995) but not in Bergström and Jarvis (1994).

To compare model validations performed with different model codes under different conditions and by different modellers, a consistent validation procedure is required and consensus regarding performance criteria must be established. An attempt to develop a standard procedure was presented in FOCUS (1995) but has not yet been tested in practice and is therefore a theoretical framework that remains to be made operational.

The primary aim of the present modeling study was to apply and evaluate the rigorous validation procedure recommended by FOCUS (1995) on the basis of laboratory and lysimeter experiments. A secondary aim was to compare the capability of four pesticide leaching models to predict pesticide leaching in a macroporous sandy loam soil.

## MATERIALS AND METHODS

### Model Codes

The four model codes used in this study were PELMO version 2.01 (Klein, 1993, 1995), PESTLA version 2.3 (Boesten and Van der Linden, 1991; Boesten, 1993), MACRO version 3.2 (Jarvis, 1991, 1994) and MIKE SHE version 5.23 (Abbot et al., 1986; Refsgaard and Storm, 1995; DHI, 1997). PELMO and PESTLA were selected because they are already used in standard registration procedures in the countries where they have been developed, Germany and the Netherlands, respectively. MACRO and MIKE SHE were selected because they seem to have the most appropriate combination of process descriptions relevant for Danish hydrological conditions, including a description of water and solute transport in macropores (Styczen and Villholth, 1994).

One major limitation of all the traditional one-dimensional pesticide leaching model codes including MACRO, PESTLA, and PELMO, is that they are restricted to describing the vertical movement of pesticides in the upper unsaturated part of the soil. Lateral transport processes occurring in the saturated zone below the root zone are not considered. MIKE SHE on the other hand contains a dynamic coupling of a one-dimensional unsaturated zone description and a three-dimensional groundwater description, including the effects of fractures on solute transport.

### Approaches to Modeling Water Flow

A precondition for modeling solute transport is an appropriate description of water flow. Different approaches are

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used in the selected model codes. In PESTLA and PELMO. preferential flow processes are not considered, assuming that water flow occurs in a uniform porous medium. The approach applied in PESTLA is the physically based description of water flow represented by Richards' equation. This requires input of the hydraulic conductivity function $K(\psi)$ and the water retention function $\theta(\psi)$. Solving the equation numerically involves iterative matrix solvers. PELMO uses a simpler "tipping bucket" method by which a continuous water balance for each computational layer is based on its water storage capacity and an empirical drainage rule. MACRO and MIKE SHE both contain process descriptions aiming to simulate preferential flow processes. In both model codes water flow is simulated using Richards' equation in the matrix and Darcian flow with a unit gradient in the macropores, but specific features like the conceptual determination of macroporosity and the process governing water exchange between matrix and macropores are different.

## Approaches to Modeling Solute Transport and Transformation

MACRO, PESTLA, and MIKE SHE apply the convection-dispersion equation, whereas PELMO calculates convective transport where dispersion only occur through the numerical solution of the equation. Representation of pesticide sorption and degradation, are more or less similar in the four model codes with at least linear equilibrium sorption and first-order decay described as a function of soil temperature and soil moisture content. All four model codes also include a description of plant uptake as passive transport with the transpiration stream. In addition to these basic processes, all model codes except MACRO include some additional options for describing reactive pesticide processes: Both PELMO, PESTLA, and MIKE SHE have Freundlich isotherms while MIKE SHE additionally contain the Langmuir isotherm and a hysteric sorption/desorption description. PELMO additionally allows increased sorption time, soil pH dependent sorption and volatilization of pesticide governed by Henry's Law.

## Experimental Background

Two types of experimental data sets were used in the validation procedure:

1. A laboratory study using large undisturbed columns (LUC) investigating steady-state water and solute transport in a 1 m deep intact soil column with a surface area of 0.196 $m^{+2}$. Sampling of the soil columns was performed by isolating a 1.5 m deep soil block with a surface area of 4 $m^{+2}$. A 0.5 m diam. cutting shoe was attached to a 1 m long steel cylinder and placed on the top of the block. Soil material around the cutting shoe was removed manually and the cylinder and shoe were allowed to fall under their own weight. Thereafter the cylinder was removed and a rubber membrane was pulled over the column and liquid rubber was poured into the annular space between the membrane and the sample. A steel mantle was tightened around each sample. The ends of each column were stabilized with rigid plastic end caps before transportation to the laboratory. In the laboratory stainless steel screens and a steel end cap were attached to the column base, and a cooling system was built around each sample (Fig. 1). Steady-state water flow through the column was maintained by a peristaltic pump connected to the effluent line from the bottom of the soil core. Further details on the experimental methodology are described in Jensen et al. (1998) and Jørgensen et al. (1998a). Prior to solute application the soil column was drained for 10 d. Thereafter steady flow of 2.75 mm $h^{-1}$ was applied beginning with a solute pulse corresponding to 2 kg a.i. $ha^{-1}$. Solutes applied were pentafluorobenzoic acid (PFBA) as a tracer and the herbicide 2-[(4-chlor-*o*-tolyl)oxy]propionic acid (MCPP). Outflow from the column was collected through time, stored at 4°C in Duran bottles with Teflon caps, and analyzed for solute content using high pressure chromatography (HPLC) technique with gradient elution at a flow rate of 1 mL/min. A-eluent was prepared by dilution of the content of one PIC-A low UV reagent bottle in 1 L of Millipore filtrated water and by adding 300 mL acetonitrile. B-eluant was pure acetonitrile. The eluants were filtered through 0.22 μm Millipore filter and degassed during eluation with Helium sparging.

2. A field lysimeter study investigating the behavior of water and solutes under natural field conditions (lysimeter dimensions: surface area of 0.707 by 0.707 m and 1.1 m deep). Two undisturbed soils cores were sampled using the same method as described for the LUC samples. Without using machine force, the soil blocks were cut out with a cutting edge placed inside the lysimeter vat leaving an annulus of approximately 7 mm between the vat and the soil core which was afterwards filled in with liquid rubber. The soil cores were installed in lysimeter metal boxes with a reservoir at the bottom for sampling of leachate (Fig. 2). Winter wheat (*Triticum aestivum* L. spp. *vulgare*) was grown in the lysimeters and around them. On 11 May 1994 the following solutes were applied: 15 kg $ha^{-1}$ potassium bromide (KBr) as a tracer and 2 kg a.i. $ha^{-1}$ of the herbicide MCPP as formulated $C^{+14}$ labeled K-salt. After the solute application, a precipitation rich period was mimicked by adding irrigation water corresponding to the precipitation at the Askov research station in May and June 1981, approximately 88 and 164 mm, respectively. During the remaining period, the lysimeters were irrigated according to average precipitation at Askov (1931–1960) which is approximately 800 mm per year. Potential evapotranspiration was calculated using Makkink (1957). Soil moisture conditions

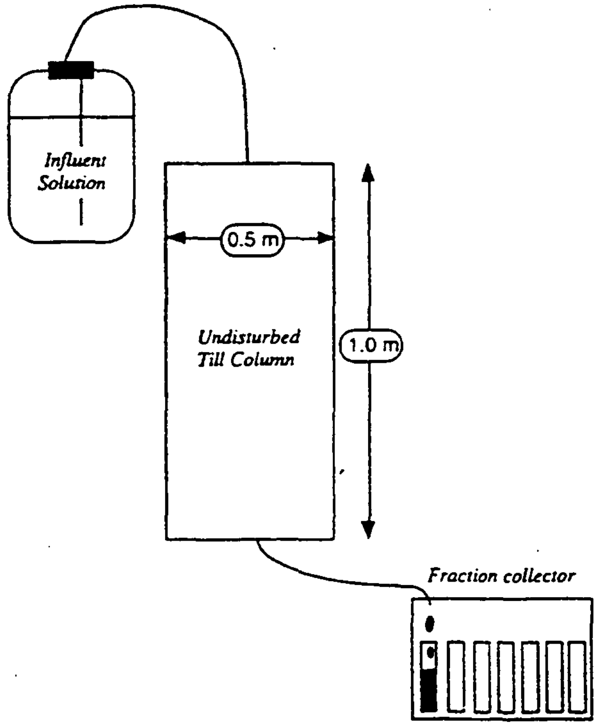


**Fig. 1. Vertical cross-section of the experimental setup of large undisturbed columns (LUC) in laboratory.**

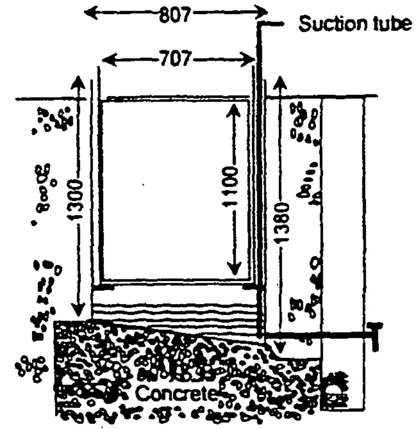


**Fig. 2.** Vertical cross-section of a lysimeter. Measurements are in millimeters.

were measured weekly at three depths (10, 30, and 50 cm) using time domain reflectometry (TDR) and soil temperature was measured hourly at two depths (10 and 30 cm)

A sandy loam soil (USDA, 1951) developed from clay-rich glacial till located in central Jutland, Denmark was used for the experiments. Water retention characteristics were determined using sandbox equipment and pressure chambers (Schjønning, 1985) and saturated hydraulic conductivities were determined in laboratory by the constant head method (Rasmussen, 1976). Unsaturated hydraulic conductivities were determined by parameter estimation from "one-step outflow" experiments (Kool et al., 1985) modified according to Jacobsen (1992). Soil physical characteristics are shown in Table 1. Pesticide properties were determined by standard laboratory methods. Linear sorption coefficients were measured at three depths according to Organization for Economic Co-operation and Development (OECD) guideline no. 106, and first-order degradation rates were determined in undisturbed soil cores at 10°C and 50% water-holding capacity. The measured MCPP properties are shown in Table 2. A more detailed description of the experiments is found in the project report (Jørgensen et al., 1998a).

## Validation Procedure

The aim of the applied stepwise approach is to determine the level of information necessary to ensure adequate simulation of pesticide behavior. The level of information refers to the quality and quantity of experimentally determined input parameters and the type and quality of measurements from leaching experiments. "Adequate simulations" refers to simulation results which fulfill some predefined success criteria based on appropriate statistical tests for the type of results that are being evaluated.

The validation procedure applied in the present study is

**Table 2.** Laboratory measured pesticide properties (MCPP).

| Depth | $K_d$ | $K_{om}$ | $T_{1/2}$, 10°C |
|---|---|---|---|
| cm | cm³ g⁻¹ | | d |
| 0–15 | 0.56 | 21 | 4.5 |
| 40–55 | 0.27 | 90 | 28.5 |
| 95–105 | 0.38 | 190 | 114 |

based on the recommendations of FOCUS (1995) which involve two sets of experimental data sets, less and more complex. The philosophy behind this procedure is to evaluate model validations performed under controlled and less complex experimental conditions relative to those under more complex conditions. As the basis for the primary model calibrations, the proposed testing procedures to evaluate different parts of the models (water transport, solute transport, and solute transformation) use data from the less complex experiment. In the present study, the less complex scenario is represented by the laboratory experiment and the more complex scenario is represented by the lysimeter experiment. The applied procedure include the four steps described in Table 3. Only three of the validation steps (2a, 3, and 4a) involve calibration and none of the steps involve calibration of pesticide sorption or degradation parameters on the basis of data from the lysimeter experiment.

## Conceptual Model Set Up

### Large Undisturbed Columns Experiment

The simulated soil column was divided into five horizons. The computational discretization varied among models, although a maximum layer depth of 5 cm was selected where possible. The lower boundary conditions differed slightly between the models. In PELMO, the flux out of the column occurs when the specified moisture capacity in the bottom compartment is exceeded. In PESTLA and MIKE SHE, the flux was determined by a unit gradient and the hydraulic conductivity in the bottom layer. In MACRO, a lysimeter boundary was selected allowing water flow only at positive pressures. The simulation period was 30 d, starting with 10 d draining from saturated conditions. Thereafter, steady flow (2.75 mm h$^{-1}$) was applied. Bare soil conditions were assumed with potential evapotranspiration equal to zero and a constant air and soil temperature at 2°C. PFBA and MCPP were applied during a 54 min pulse after 10 d of free drainage.

### Lysimeter Experiment

The simulated soil column was divided into five horizons. The computational discretization and boundary conditions were similar to those for the LUC experiment. The simulation period was 28 Apr. 1994 to 31 Dec. 1995 with input of daily climatic data as driving variables. Crop parameters were selected to meet the observed crop development with regard to

**Table 1.** Soil physical characteristics. $K_s$ was used as total saturated hydraulic conductivity (matrix + macropores) in all models and $K_b$ was used as the hydraulic conductivity close to matrix saturation in the two models describing preferential flow.

| Depth | Clay <2 µm | Silt 2–20 µm | Coarse silt 20–63 µm | Sand | Bulk density | $\theta_s$ | $K_s$ | $K_b$ | Organic matter |
|---|---|---|---|---|---|---|---|---|---|
| | % | | | | g cm⁻³ | – | 10⁻⁶ ms⁻¹ | | % |
| 10–13 | 14.7 | 13.4 | 12.5 | 59.4 | 1.63 | 0.39 | 0.84 | 0.14 | 2.6 |
| 25–28 | 13.7 | 13.4 | 16.0 | 56.9 | 1.64 | 0.38 | 4.50 | 0.06 | 1.5 |
| 30–33 | 12.7 | 13.4 | 17.8 | 56.1 | 1.58 | 0.40 | 11.87 | 0.19 | 0.9 |
| 50–53 | 14.7 | 14.4 | 24.6 | 46.3 | 1.64 | 0.38 | 13.39 | 0.69 | 0.3 |
| 90–93 | 17.6 | 15.5 | 25.9 | 41.0 | 1.82 | 0.31 | – | – | 0.2 |

$K_s$ = measured saturated hydraulic conductivity, $K_b$ = saturated hydraulic conductivity estimated from the "one-step outflow" method, $\theta_s$ = porosity.

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Table 3. Overview of the applied stepwise validation procedure.

| Step | | Modeling procedure |
|---|---|---|
| 1 | a | *Blind simulation* on LUC data, evaluation of predicted water and MCPP behavior. |
| | b | *Blind simulation* on lysimeter data, evaluation of predicted water and MCPP behavior. |
| 2 | a | *Calibration* performed on measured LUC data in terms of water and PFBA fluxes. Evaluation of predicted MCPP behavior. |
| | b | *Simulation* of the lysimeter experiment using the calibrated parameters from the calibration on LUC data (step 2a). Evaluation of predicted water and MCPP behavior. |
| 3 | | *Calibration* performed on measured lysimeter data in terms of percolation and Br behavior. Evaluation of predicted MCPP behavior. |
| 4 | a | *Calibration* of MCPP behavior performed on measured MCPP data from the LUC experiment. |
| | b1 | *Simulation* of the lysimeter experiment using calibrated parameters from the LUC simulations in steps 2a and 4a. Evaluation of predicted MCPP behavior. |
| | b2 | *Simulation* of the lysimeter experiments using calibrated parameters from steps 3 and 4a, meaning that water and solute transport is calibrated using the lysimeter measurements of water and Br behavior whereas pesticide parameters are calibrated using LUC measurements of MCPP. leaching. |

† LUC = large undisturbed columns.

leaf area index and solutes were applied according to experimental conditions.

The initial blind parameterization of all model codes was as much as possible based on measured parameters, but without any information obtained from the flow and transport experiments themselves. However, due to differences between the model codes in terms of mathematical representation of the described reactions and due to lack of measured parameters, the initial setups contained a certain degree of subjective assessments. For example, the measured soil physical characteristics such as retention curves and hydraulic conductivity functions could only be used directly in some model codes, and were therefore fitted to empirical functions. In PELMO only three values are used to describe the hydrological conditions in terms of the field capacity value for each soil horizon. In MACRO, the retention curves have to be fitted to the Brooks and Corey (1964) equation which is best suited for sandy soils (Styczen and Villholth, 1994), and the hydraulic conductivity function has to follow the Mualem (1976) approach. In PESTLA two options for input of both retention curves and hydraulic conductivity functions are available as the data can be given directly in tabular form, or as parameters fitted to the van Genuchten (1980) equation. However, as the input structure required for the tables is time consuming when calibrating, the van Genuchten parameters were used in these simulations. In MIKE SHE, the retention curves are given directly as tables, whereas the hydraulic conductivity function must be fitted to an exponential function. These differences in the parameterization of the hydrological properties caused differences in the simulated flow patterns even between model codes using the same flow equations.

Some of the parameters describing macropore flow in MACRO and MIKE SHE are not measurable and values were therefore initially chosen as "educated guesses" and used as calibration parameters. These include an empirical parameter governing the transport between matrix and macropores and the boundary threshold value boundary water content (MACRO) or threshold pressure (MIKE SHE) determining the moisture conditions at which macropore flow is initiated. Both parameters, which have significant influence on the simulation results, are not directly measurable at present though they may be conceptually determined from other soil physical characteristics such as retention curves or soil structure.

Other examples of subjective parameter assessments are the description of evapotranspiration which differs among all four model codes in terms of the number and type of required input data and the extent of plant uptake of the simulated solutes (PFBA, Bromide and MCPP), which is not well known. This latter process was neglected in the initial parameterizations and used as a calibration parameter in the latter stages.

A practical problem with model setup for PESTLA, MACRO, and MIKE SHE was the virtual presence of overland flow when the infiltration capacity was exceeded. This was especially a problem when simulating the LUC experiments, as the flow rate here was very high (2.75 mm h$^{-1}$). Increasing the hydraulic conductivity ($K_s$) in the top soil layer allowed complete infiltration of water.

### Statistics and Evaluation Criteria

Model performance was evaluated with respect to their use for registration purposes, that is, the ability to predict pesticide concentrations in groundwater aquifers. Using only one-dimensional leaching models this effort is restricted to evaluation of the average annual concentration leaving the root zone (1–1.5 m depth) based on leached pesticide mass and water discharge. Additionally, the magnitude of the peak concentration may be of interest for estimation of potential fluctuations in groundwater concentrations.

For each step in the validation procedure the following simulation results were stored and evaluated by graphical and statistical means: accumulated percolation, accumulated pesticide leaching, and a time series of pesticide concentrations from simulations of both LUC- and lysimeter experiments and an additional time series of soil moisture contents at three depths, and soil temperatures at two depths from the lysimeter simulations. To ensure consistent evaluation of model performances, statistical tests and success criteria were selected based on suggestions from FOCUS (1995) (see Appendix).

## RESULTS AND DISCUSSION

The simulations presented in this section summarizes results illustrating the value of the validation procedure suggested by FOCUS and the performance of the four models tested. Complete description of the validation progress is given in Jørgensen et al. (1998a). The main objective of applying the described validation procedure was to determine the information necessary to make the models describe the most "natural" situation which in this study is represented by the lysimeter data. The discussion therefore focuses on descriptions of water and solute behavior in the lysimeters. Hence simulation results from the LUC data corresponding to steps 1a, 2a, and 4a are not shown. In both LUC and lysimeter experiments, preferential flow in the macropores controlled to a very high extent flow and solute transport in the soil. Significant solute breakthrough occurred

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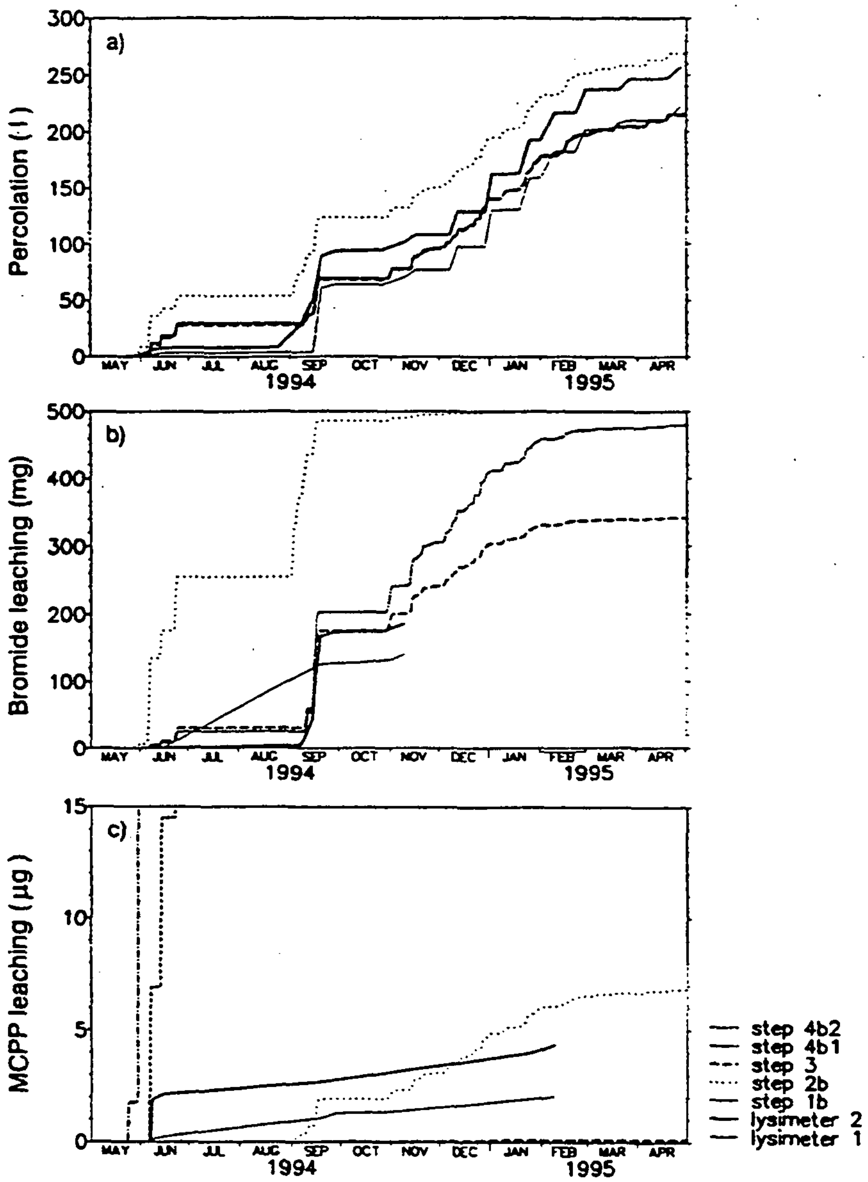


Fig. 3. PELMO. Summary of stepwise validation runs. (*a*) accumulated percolation, (*b*) accumulated Br leaching, (*c*) accumulated MCPP leaching.

readily at much less water percolation than 1 pore volume. This experimental result implies that correct prediction of pesticide transport in the soil is highly sensitive to an accurate description of the physical flow system.

Examples of the stepwise results from simulations of the lysimeter experiments using PELMO and MIKE SHE are shown in Fig. 3 and 4. The total accumulated percolation is already described well by both models in the blind simulation (step 1b) though they both overestimate percolation in early June 1994 prior to solute breakthrough (Fig. 3a and 4a). This problem, which was also observed in the simulations with MACRO (not shown), may be a combination of uncertainty regarding the initial soil moisture conditions, inadequate description of the evapotranspiration process in some of the model codes and a possible "clothesline" effect which may have caused enhanced evapotranspiration due to dry areas surrounding the irrigated lysimeters during the spring period. This latter phenomenon has been observed in similar lysimeter studies (Boesten, 1994). The accumulated Br leaching (Fig. 3b and 4b) from the blind simulations was best represented with PELMO. The accumulated MCPP leaching (Fig. 3c and 4c) was best represented by MIKE SHE, while virtually no leaching was simulated by PELMO. This difference between the predicted leaching of conservative and reactive solutes by the two models is due to the combined effect of transport velocity and rate of the sorption process as PELMO does not allow for preferential transport which in MIKE SHE automatically diminishes the effect of the applied linear equilibrium sorption on MCPP leaching.

In step 2b, water and solute behavior in the lysimeter experiments were simulated using calibrated hydrological parameters obtained in step 2a. With PELMO an

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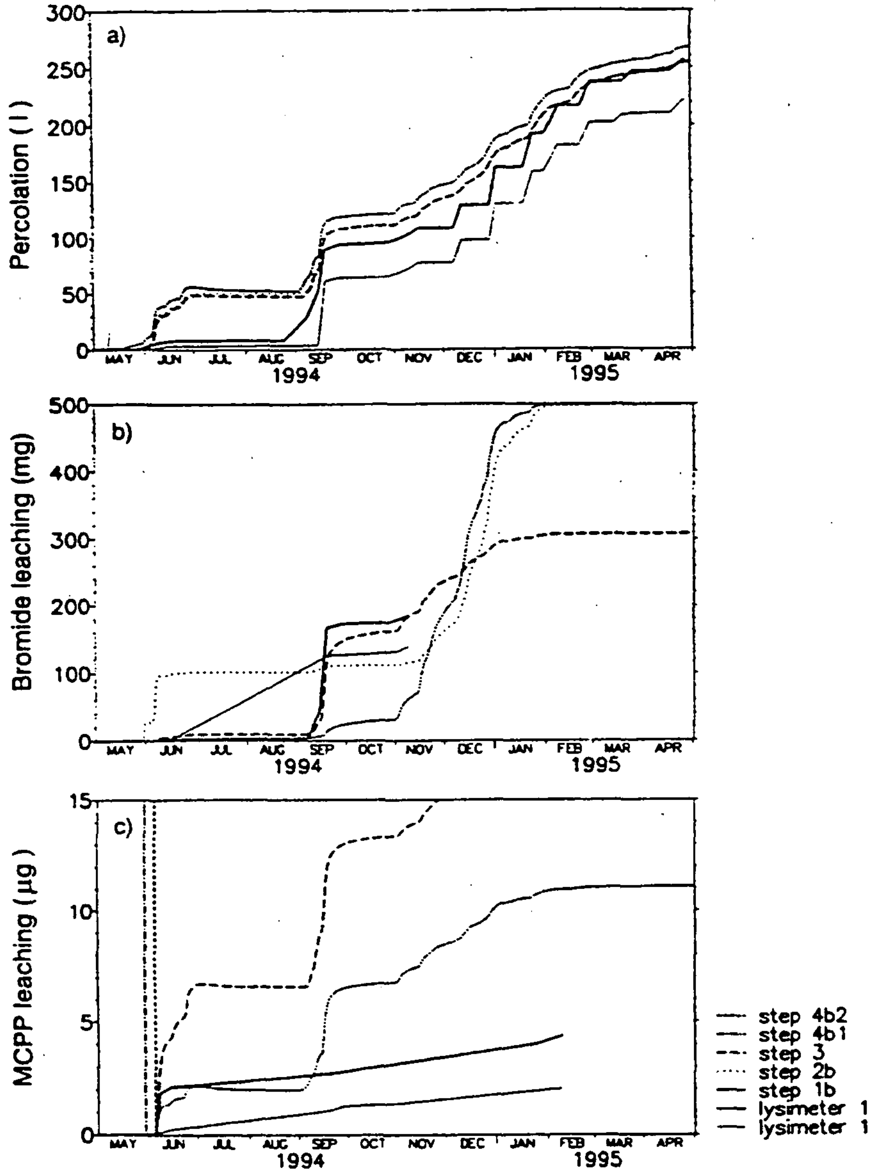


**Fig. 4. MIKE SHE. Summary of stepwise validation runs. (*a*) accumulated percolation, (*b*) accumulated Br leaching, (*c*) accumulated MCPP leaching.**

improvement of the predicted PFBA behavior in the LUC experiments could only be obtained by violating the physical description of the soil column by decreasing the soil moisture content at field capacity significantly to resemble a situation with mobile and immobile water. These adjustments improved the predicted MCPP leaching significantly (Fig. 3c) but in turn resulted in less accurate predictions of drainage fluxes and Br breakthrough (Fig. 3a and 3b—step 2b). In MIKE SHE, calibration of PFBA behavior in step 2a was conducted by adjusting the distributed representation of parameters describing mass transfer between matrix and macropores. This, however, did not improve prediction of either water, Br or MCPP behavior for the lysimeter experiment (Fig. 4—step 2b). One reason for this is the overestimated percolation prior to solute breakthrough at 8 June. When adjusting the mass transfer parameters, solute diffusion between domains became more dominating in the upper soil layers than was the case in the blind simulations which resulted in too early solute breakthrough in too much water. Based on this result it is difficult to evaluate the value of the calibrated mass transfer parameters.

In step 3, calibrations were performed on water and Br behavior using the measured lysimeter data. In both PELMO and MIKE SHE this was conducted by introducing plant uptake of solutes and fitting of the dispersion parameters, with additional adjustments of the retention curves and the representation of preferential flow for MIKE SHE. This improved the simulation of water and Br behavior for both PELMO and MIKE SHE (Fig. 3a+b and 4a+b—step 3). The simulation of MCPP behavior was not significantly improved by these adjustments (Fig. 3c and 4c— step 3). PELMO still did

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not simulate much leaching and in MIKE SHE the adjustments caused further overestimation of MCPP leaching due to decreased porosities from the adjustment of the retention curves.

In steps 4b1 and 4b2 where calibration was performed using measured MCPP behavior from the LUC experiment (step 4a, not shown), both models strongly overestimated MCPP leaching in these steps (Fig. 3c and 4c—steps 4b1 and 4b2). The calibrations were conducted by decreasing the sorption coefficients significantly to meet the early breakthrough observed in the LUC experiment.

## Evaluation of Model Performance

All four models already predicted the accumulated percolation well in the blind simulations (Fig. 5a) although MIKE SHE overestimated it. The predicted percolations were only moderately improved for all models after calibrations were performed in step 3. The simulated Br behavior in the blind simulation (Fig. 5b) on the other hand varied significantly between the models. MACRO and MIKE SHE underestimated the accumulated Br leaching in September 1994 and PELMO and PESTLA overestimated it.

After calibrations in step 3 all models predicted Br leaching well with regard to both timing and quantity. Predictions of MCPP behavior showed larger differences among the models. In the blind simulation (Fig. 5c) significant MCPP leaching was only predicted by the two models including a description of preferential flow, that is, MACRO and MIKE SHE. For PELMO and PESTLA, the linear equilibrium sorption process used in combination with homogeneous vertical water flow resulted in more MCPP retardation and less transport.

## Statistical Summary

For each step in the validation procedure statistical tests were applied for simulated results of accumulated

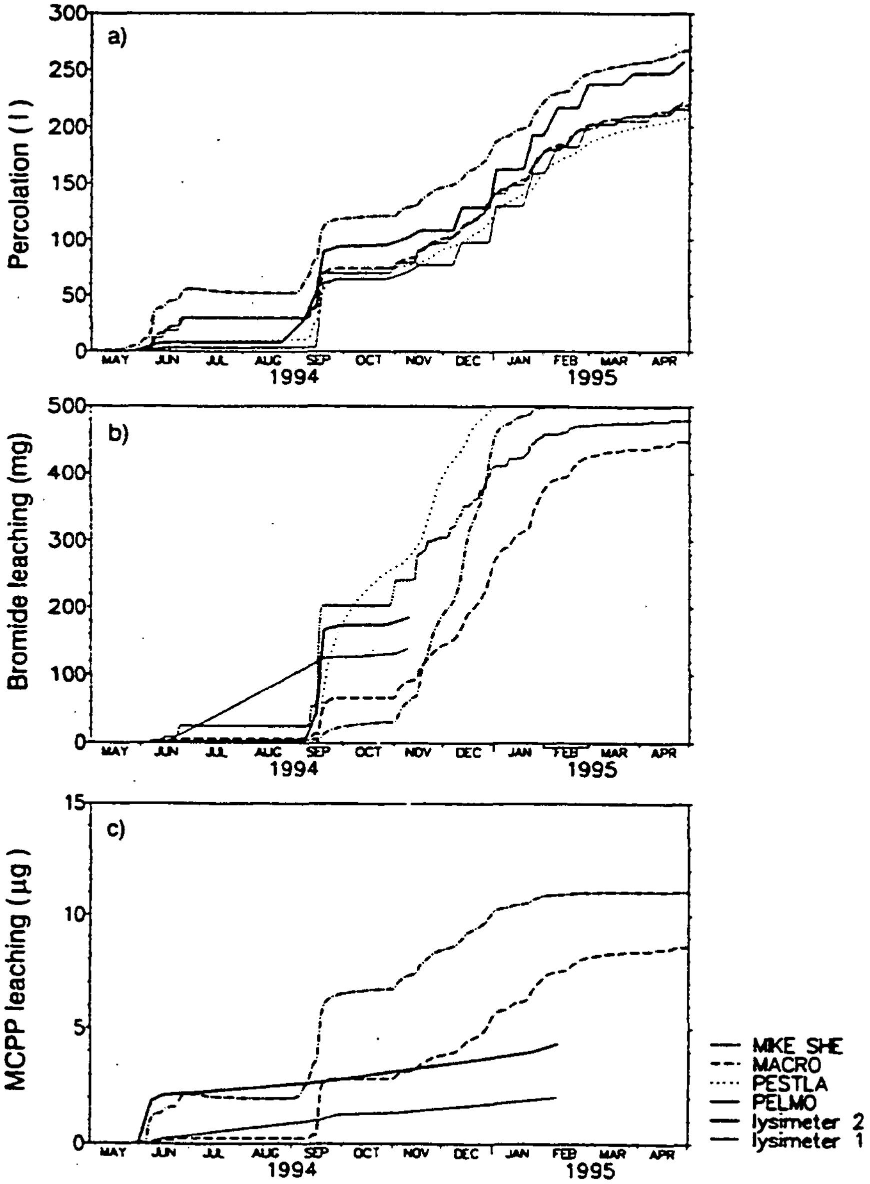


**Fig. 5.** Blind simulation (step 1b). Comparison of model performances. (*a*) accumulated percolation, (*b*) accumulated Br leaching, (*c*) accumulated MCPP leaching.

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percolation, accumulated MCPP leaching and the magnitude of the peak concentration. An overview of the statistical results from the simulation of the lysimeter experiments (steps 1b, 2b, 3, 4b1, and 4b2) are shown in Table 4.

The total accumulated percolation from the lysimeter experiment was described reasonably well by all models, although the success criterion of 8% deviation from the central estimate was generally only met after model calibration. However, as the applied criterion does not account for the variation among lysimeters, which in this study corresponds to a 15% range, one could argue that the selected criterion is rather strict as compared to the criteria applied to the pesticide data which implicitly take lysimeter variation into account. This may also apply to the simulation of soil moisture content and soil temperature where neither of the models met the applied success criteria which is also based on the central estimate of measurements. The MCPP behavior was simulated very well by MACRO and MIKE SHE which both met the applied success criterion in the blind simulation (1b) without any calibration. PELMO reached the criterion in step 2b and PESTLA in step 4b1. Separate evaluation of the simulations for the two types of experiments revealed a similar picture of the importance of model ability to describe preferential flow. Thus in both experiments MACRO and MIKE SHE met the applied success criteria earlier in the validation procedure than PELMO and PESTLA did.

Naturally, statistics and evaluation criteria influences the conclusions with regard to model performance hence caution must be taken in their selection. A measurement interval corresponding to a coefficient of variation of approximately 50% was used for simulated MCPP behavior. If for example the exact measured interval had been used instead (obs1; obs2), the conclusion would have differed with respect to the step at which the models qualified to simulate the simulated peak concentration but not with respect to the accumulated leaching. For comparative modeling studies it is crucial that model performance is evaluated using objective criteria.

## CONCLUSION

The main conclusion regarding the validation procedure is that the stepwise approach is a valuable tool for evaluating the performance of different model components and ensuring consistent model comparisons. Thus, the validation procedure proposed by FOCUS (1995) provides a useful framework for rigorous and objective assessment of model performance. Such validation tests should always be carried out before a model is used for registration or other purposes.

Another conclusion is that the two experiments used in the procedure should have more similar conditions than was the case in this study to benefit from all the steps in the validation procedure. The steady-state situation with high flow velocity in the LUC experiment and the dynamic situation in the lysimeter experiment represented two different situations with regard to the solute movement in time. Large undisturbed column data, were valuable by contributing to the description of the physical flow system and the influence of the natural soil structure on preferential flow and solute transport. This description was consistent with results of the lysimeter experiments.

The ultimate goal of model validation is to evaluate the ability to predict pesticide behavior under natural field conditions. In this context, artificial lower boundary conditions in lysimeter experiments may cause biases. In the validation procedure proposed in FOCUS (1995) this aspect is addressed by using lysimeter experiments as the less complex situation and field experi-

**Table 4. Evaluation of model performance from the lysimeter simulations. Underscore indicate the validation step, if any, at which the success criteria suggested by FOCUS is met for each particular data type. "neg." indicates that the factor of $f$ test was not successful at either $f = 2$ or $f = 5$.**

| Model | | Percolation, L | MCPP leaching, µg | MCPP peak conc., µg $L^{-1}$ | Soil moisture content (−) | Soil temperature, °C |
|---|---|---|---|---|---|---|
| Statistical test | | % Dev. | $f$-test | $f$-test | RMSE† | RMSE |
| Success criteria | | 8% | $f = 2$ | $f = 2$ | <0.1 | <0.1 |
| PELMO | 1b | 10 | Neg. | Neg. | 0.45 | 0.125 |
| | 2b | 12 | $\underline{f = 2}$ | $\underline{f = 2}$ | 0.47 | – |
| | 3 | 10 | Neg. | Neg. | 0.29 | – |
| | 4b1 | – | Neg. | Neg. | – | – |
| | 4b2 | – | Neg. | $f = 5$ | – | – |
| PESTLA | 1b | 13 | Neg. | Neg. | 0.52 | 0.145 |
| | 2b | 13 | Neg. | Neg. | 0.52 | – |
| | 3 | $\underline{6}$ | Neg. | Neg. | 0.31 | – |
| | 4b1 | – | $\underline{f = 2}$ | $\underline{f = 2}$ | – | – |
| | 4b2 | – | $f = 2$ | $f = 2$ | – | – |
| MACRO | 1b | $\underline{8}$ | $f = 2$ | $f = 2$ | 0.40 | 0.144 |
| | 2b | 3 | $f = 5$ | $f = 5$ | 0.33 | – |
| | 3 | 2 | $f = 2$ | $f = 2$ | 0.32 | – |
| | 4b1 | – | $f = 5$ | $f = 2$ | – | – |
| | 4b2 | – | Neg. | $f = 2$ | – | – |
| MIKE SHE | 1b | 12 | $\underline{f = 2}$ | $\underline{f = 2}$ | 0.51 | 0.112 |
| | 2b | $\underline{6}$ | Neg. | Neg. | 0.54 | – |
| | 3 | 7 | $f = 5$ | $f = 2$ | 0.35 | – |
| | 4b1 | – | Neg. | Neg. | – | – |
| | 4b2 | – | Neg. | Neg. | – | – |

† RMSE = root mean square error.

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ments including natural lower boundary conditions as the more complex condition. To fulfill the ultimate goal for validation the procedure applied in the present study should therefore ideally be extended to also include a field study. Another objective of the applied validation procedure is to test whether pesticide properties measured in the laboratory are representative for the dynamic conditions found in nature. In the present study measured sorption and degradation properties were site specific and the assumption of representativeness of measured values cannot be rejected as both of the models accounting for preferential flow already qualified in the blind simulation.

The main result regarding model performances is that model codes containing a description of preferential flow (MACRO and MIKE SHE) passed the validation criteria for pesticide fluxes already in the blind step of the validation procedure whereas PELMO and PESTLA needed calibration to mimic the influence of preferential flow processes. Used as predictive tools aiming to provide realistic concentrations actually expected to be found in the groundwater and therefore directly compared to legislative concentration limits, the modeling tool must represent actual situations as close as possible using state-of-the-art knowledge, that is, by including preferential flow processes. Otherwise intercomparative rankings of pesticides using simple mobility indexes based on pesticide properties only or steady-state transport models (Jury et al., 1987) could be adequate.

A major complication regarding modeling of macropore behavior, however, is that parameterization is difficult. Soil-related parameters for use in MACRO and MIKE SHE are at present difficult to measure and, in addition, the extent and distribution of preferential flow pattern in the root zone will also be influenced by the agricultural management practice, for example, due to different tillage operations (Andreini and Steenhuis, 1990; Petersen et al., 1996) or crop types (Caron et al., 1996). Hence a standard macroporous setup will require thorough testing and validation.


## ACKNOWLEDGMENTS

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## APPENDIX

### Statistical Tests

Three types of statistical tests were applied to simulation results obtained in the four validation steps:

1. For evaluation of the overall fit of simulated time series of soil moisture contents and soil temperatures the root mean square error (RMSE) using the measured average between the two lysimeters was applied (e.g., Walker et al., 1995). The success criterion was RMSE < 0.1.
2. For evaluation of simulated accumulated water discharge the percentage deviation from the measured average between the two lysimeters was used. The success criterion was ≤ |8%|.
3. For evaluation of simulated accumulated pesticide leaching and pesticide peak concentrations the factor of $f$ approach suggested by Parrish and Smith (1990) and further developed by Boekhold et al. (1993) was applied. This type of calculation is a comparison between simulated and observed values taking the observed variation in measured data into account. $\sigma$ is the standard deviation of the log transformed data.

$$\left(10^{(\overline{\log(O_j)}-n\sigma)}, 10^{(\overline{\log(O_j)}-n\sigma)}\right)_i \textit{ compared against } \frac{P_i}{f}, P_i \times f \qquad [1]$$

$O =$ the observed value
$P =$ the predicted value
$i =$ number of data pairs (observed, predicted) for comparison
$j =$ number of replicate measurements
$n$ equals 1 or 2
$f$ is usually on the order of 2 to 5

Comparing the calculated uncertainty intervals for the observed and simulated values provides an opportunity to evaluate the hypothesis of the model simulations being valid or rejected at different levels, for example, $f = 2$ or $f = 5$.

The factor of $f$ test was originally developed for studies including several replicate measurements giving the possibility of estimating standard deviations and uncertainty intervals on the observed data. However, in this study such statistics could not be applied directly as the lysimeter data were based on only two replicate experiments. The uncertainty intervals of the observed data used for the factor of $f$ tests were therefore estimated based on the observed difference between the two replicate measurements. The applied approach takes into account the measured variation between the two lysimeters using an estimated "standard deviation" corresponding to a coefficient of variation close to 50%. The constructed logarithmic standard deviation was calculated as "$\tau$" = | log(O) − average log(O) | where O = observed values. The assumed uncertainty interval on the observed values was then calculated using the equation described for the factor of $f$ test with $n = 2$:

$$O \in \left(10^{(\text{av.}\log(O)-n\text{“}\tau\text{”})}\ 10^{(\text{av.}\log(O)+n\text{“}\tau\text{”})}\right) \qquad [2]$$

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# An Integrated Model for the Danubian Lowland – Methodology and Applications

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**Abstract.** A unique integrated modelling system has been developed and applied for environmental assessment studies in connection with the Gabcikovo hydropower scheme along the Danube. The modelling system integrates model codes for describing the reservoir (2D flow, eutrophication, sediment transport), the river and river branches (1D flow including effects of hydraulic control structures, water quality, sediment transport), the ground water (3D flow, solute transport, geochemistry), agricultural aspects (crop yield, irrigation, nitrogen leaching) and flood plain conditions (dynamics of inundation pattern, ground water and soil moisture conditions, and water quality). The uniqueness of the established modelling system is the integration between the individual model codes, each of which provides complex descriptions of the various processes. The validation tests have generally been carried out for the individual models, whereas only a few tests on the integrated model were possible. Based on discussion and examples, it is concluded that the results from the integrated model can be assumed less uncertain than outputs from the individual model components. In an example, the impacts of the Gabcikovo scheme on the ecologically unique wetlands created by the river branch system downstream of the new reservoir have been simulated. In this case, the impacts of alternative water management scenarios on ecologically important factors such as flood frequency and duration, depth of flooding, depth to ground water table, capillary rise, flow velocities, sedimentation and water quality in the river system have been explicitly calculated.

**Key words:** Danube, environmental impacts, floodplain, Gabcikovo, groundwater, hydropower, integrated modelling, river branch.

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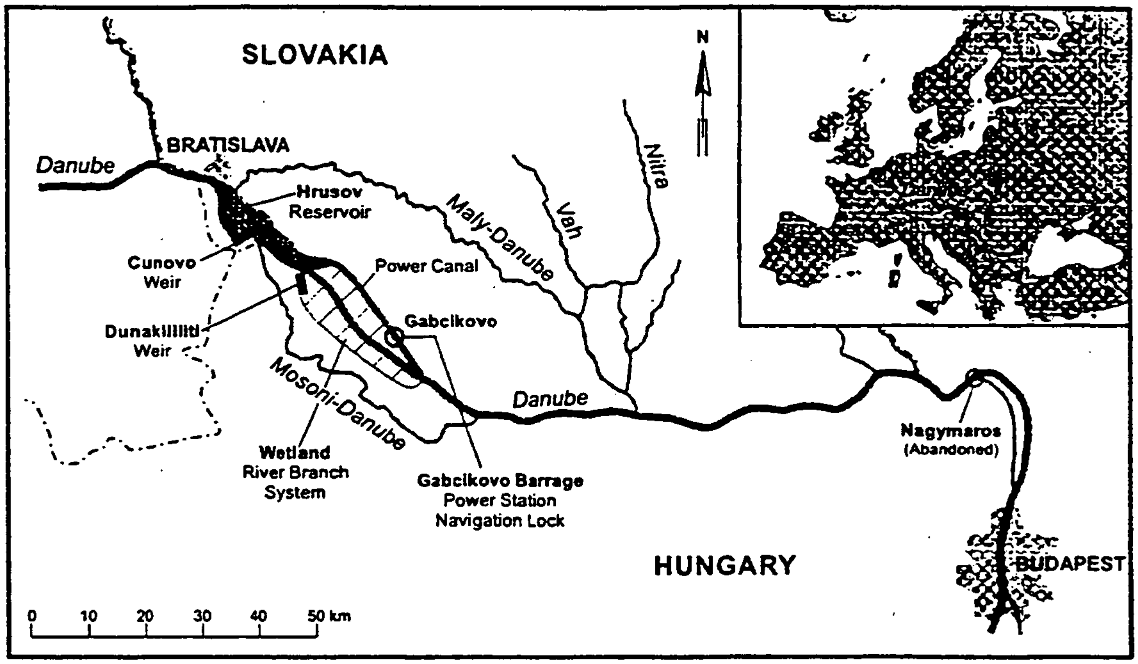


*Figure 1.* The Danubian Lowland with the new reservoir and the Gabcikovo scheme.

## 1. Introduction

### 1.1. THE DANUBIAN LOWLAND AND THE GABCIKOVO HYDROPOWER SCHEME

The Danubian Lowland (Figure 1) in Slovakia and Hungary between Bratislava and Komárno is an inland delta (an alluvial fan) formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which receives around 30 $m^3$ $s^{-1}$ infiltration water from the Danube throughout the year, in the upper parts of the area and returns it to the Danube and the drainage canals in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

Human influence has gradually changed the hydrological regime in the area. Construction of dams upstream of Bratislava together with straightening and embanking of the river for navigational and flood protection purposes as well as exploitation of river sediments have significantly deepened the river bed and lowered the water level in the river and surrounding ground water level. These changes have had a significant influence on the ground water regime as well as the sensitive riverine forests downstream of Bratislava. Despite this basically negative trend the floodplain area with its alluvial forests and associated ecosystems still represents a unique landscape of outstanding ecological importance.

The Gabcikovo hydropower scheme was put into operation in 1992. A large number of hydraulic structures has been established as part of the hydropower scheme. The key structures are a system of weirs across the Danube at Cunovo 15 km downstream of Bratislava, a reservoir created by the damming at Cunovo, a 30 km long lined power and navigation canal, outside the floodplain area, parallel to the Danube River with intake to the hydropower plant, a hydropower plant and two

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ship-locks at Gabcikovo, and an intake structure at Dobrohost, 10 km downstream of Cunovo, diverting water from the new canal to the river branch system. The entire scheme has significantly affected the hydrological regime and the ecosystem of the region, see, e.g., Mucha *et al.* (1997). The scheme was originally planned as a joint effort between former Czecho-Slovakia and Hungary, and the major parts of the construction were carried out as such on the basis of a 1977 international treaty. However, since 1989 Gabcikovo has been a major matter of controversy between Slovakia and Hungary, who have referred some disputed questions to international expert groups (EC, 1992, 1993a, b) and others to the International Court of Justice in The Hague (ICJ, 1997).

Comprehensive monitoring and assessments of environmental impacts have been made, see Mucha (1995) for an overview. Since 1995 a joint Slovak-Hungarian monitoring program has been carried out (JAR, 1995, 1996, 1997).

### 1.2. NEED FOR INTEGRATED MODELLING

The hydrological regime in the area is very dynamic with so many crucial links and feedback mechanisms between the various parts of the surface- and subsurface water regimes that integrated modelling is required to thoroughly assess environmental impacts of the hydropower scheme. This is illustrated by the following three examples:

- *Ground water quality*. Based on qualitative arguments it was hypothesised that the damming and creation of the reservoir might lead to changes in the oxidation-reduction state of the ground water. The reason for this is that the reservoir might increase infiltration from the Danube to the aquifer because of increased head gradients. On the other hand, fine sediment matter might accumulate on the reservoir bottom, thereby creating a reactive sediment layer. The river water infiltrating to the aquifer has to pass this layer, which might induce a change in the oxidation status of the infiltrating water. This could affect the quality of the ground water from being oxic or suboxic towards being anoxic, which is undesirable for Bratislava's water works, most of which are located near the reservoir. Thus, the oxidation-reduction state of the groundwater is intimately linked to a balance between the rates of infiltrating reducing water and the aquifer oxidizing capacity. The infiltrating water is linked to the hydraulic behaviour of the reservoir: how large is the infiltration area and at which rates does the infiltration take place at different locations. However, without an integrated model it is not possible to quantify whether and under which conditions these mechanisms play a significant role in practise, whether they are correct in principle but without practical importance, and what measures should be realised.
- *Agricultural production*. Changes in discharges in the Danube caused by diversion of some of the water through the power canal and creation of a reservoir

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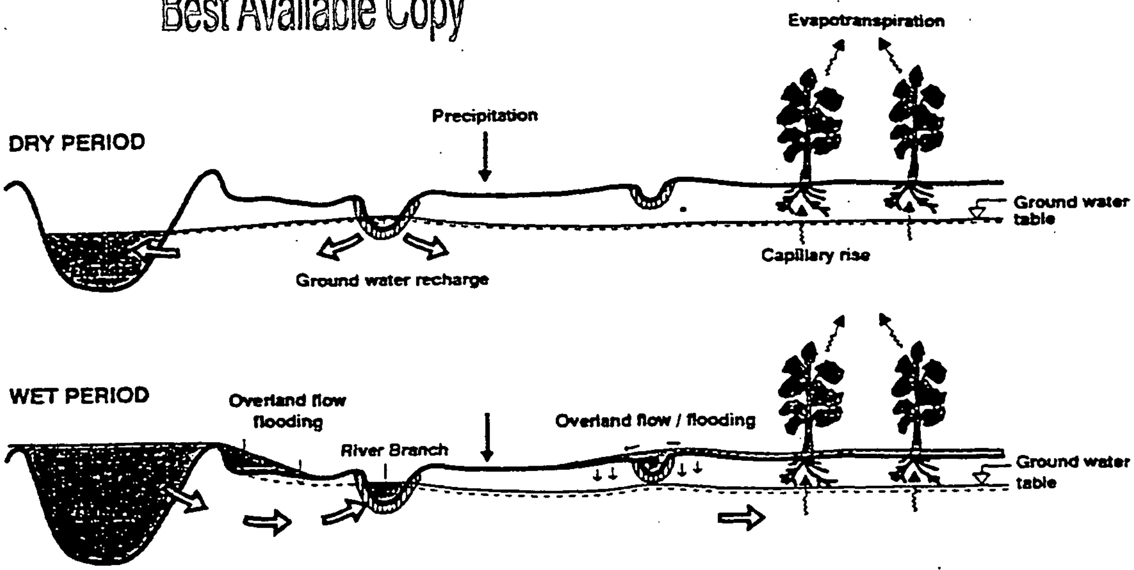


*Figure 2.* Important processes and their interactions with regard to floodplain hydrology.

would lead to changes in the ground water levels. As the agricultural crops depend on capillary rise from the shallow ground water table and irrigation, the new hydrological situation created by the damming of the Danube might influence both the crop yield, the irrigation requirements and the nitrogen leaching. Traditional crop models describing the root zone are not sufficient in this case, because the lower boundary conditions (ground water levels) are changed in a way that can only be quantified if also the reservoir, the river and canal system and the aquifer are explicitly included in the modelling.

- *Floodplain ecosystem.* The flora and fauna, which in the floodplain area are dominated by the river side branches, depend on many factors such as flooding dynamics, flow velocities, depth of ground water table, soil moisture, water quality and sediments. Also in this case the important factors depend on the interaction between the groundwater and the surface water systems (illustrated in Figure 2), and even on water quality and sediments in the surface water system, so that quantitative impact assessments require an integrated modelling approach.

## 2. Integrated Modelling System

### 2.1. INDIVIDUAL MODEL COMPONENTS

An integrated modelling system (Figure 3) has been established by combining the following existing and well proven model codes:

- *MIKE SHE* (Refsgaard and Storm, 1995) which, on a catchment scale, can simulate the major flow and transport processes in the hydrological cycle:

  – 1-D flow and transport in the unsaturated zone

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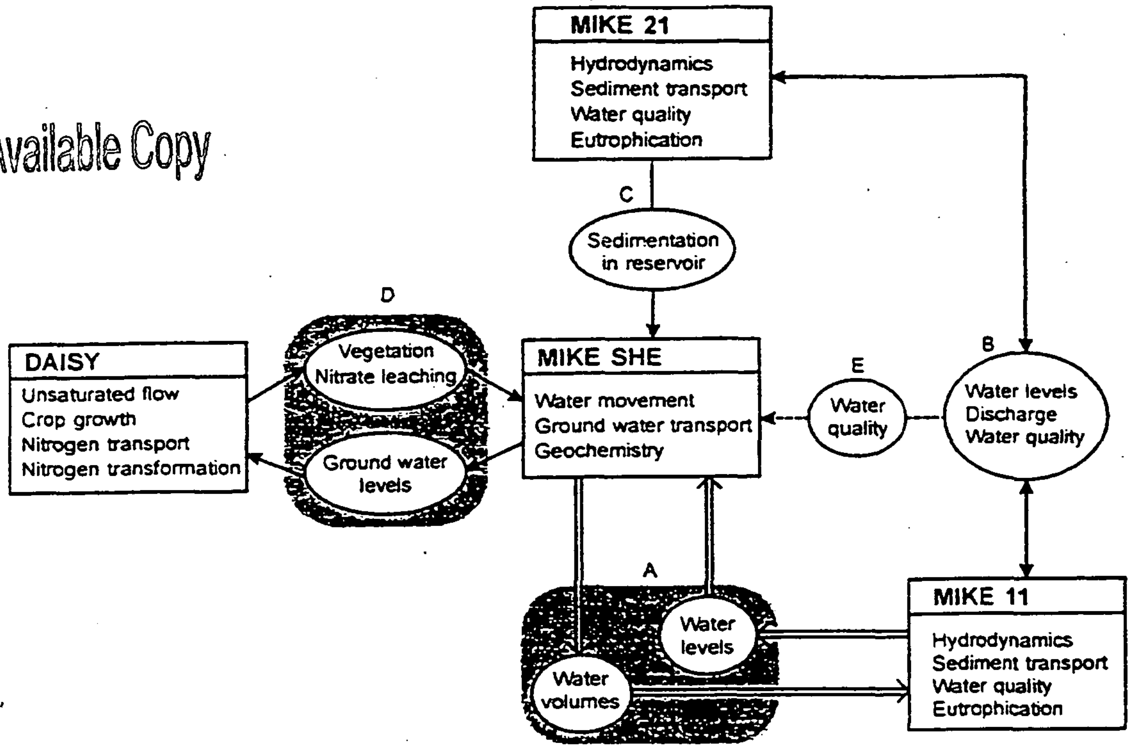


*Figure 3.* Structure of the integrated modelling system with indication of the interactions between the individual models.

- 3-D flow and transport in the ground water zone
- 2-D flow and transport on the ground surface
- 1-D flow and transport in the river.

All of the above processes are fully coupled allowing for feedback's and interactions between components. In addition, MIKE SHE includes modules for multi-component geochemical and biodegradation reactions in the saturated zone (Engesgaard, 1996).

- *MIKE 11* (Havnø *et al.*, 1995), is a one-dimensional river modelling system. MIKE 11 is used for simulating hydraulics, sediment transport and morphology, and water quality. MIKE 11 is based on the complete dynamic wave formulation of the Saint Venant equations. The modules for sediment transport and morphology are able to deal with cohesive and noncohesive sediment transport, as well as the accompanying morphological changes of the river bed. The noncohesive model operates on a number of different grain sizes.
- *MIKE 21* (DHI, 1995), which has the same basic characteristics as MIKE 11, extended to two horizontal dimensions, and is used for reservoir modelling.
- MIKE 11 and MIKE 21 include *River/Reservoir Water Quality (WQ) and Eutrophication (EU)* (Havnø *et al.*, 1995; VKI, 1995) modules to describe oxygen, ammonium, nitrate and phosphorus concentrations and oxygen demands as well as eutrophication issues such as bio-mass production and degradation.
- *DAISY* (Hansen *et al.*, 1991) is a one-dimensional root zone model for simulation of soil water dynamics, crop growth and nitrogen dynamics for various agricultural management practices and strategies.

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## 2.2. INTEGRATION OF MODEL COMPONENTS

The integrated modelling system is formed by the exchange of data and feedbacks between the individual modelling systems. The structure of the integrated modelling system and the exchange of data between the various modelling systems are illustrated in general in Figure 3 and the steps in the integrated modelling is described further in Section 6.2 and illustrated in Figure 10 for the case of flood plain modelling. The interfaces between the various models indicated in Figure 3 are

A) MIKE SHE forms the core of the integrated modelling system having interfaces to all the individual modelling systems. The coupling of MIKE SHE and MIKE 11 is a fully dynamic coupling where data is exchanged within each computational time step, see Section 2.3 below.
B) Results of eutrophication simulations with MIKE 21 in the reservoir are used to estimate the concentration of various water quality parameters in the water that enters the Danube downstream of the reservoir. This information serves as boundary conditions for water quality simulations for the Danube using MIKE 11.
C) Sediment transport simulations in the reservoir with MIKE 21 provide information on the amount of fine sediment on the bottom of the reservoir. The simulated grain size distribution and sediment layer thickness is used to calculate leakage coefficients, which are used in ground water modelling with MIKE SHE to calculate the exchange of water between the reservoir and the aquifer.
D) The DAISY model simulates vegetation parameters which are used in MIKE SHE to simulate the actual evapotranspiration. Ground water levels simulated with MIKE SHE act as lower boundary conditions for DAISY unsaturated zone simulations. Consequently, this process is iterative and requires several model simulations.
E) Results from water quality simulations with MIKE 11 and MIKE 21 provide estimates of the concentration of various components/parameters in the water that infiltrates to the aquifer from the Danube and the reservoir. This can be used in the ground water quality simulations (geochemistry) with MIKE SHE.

A general discussion on the limitations in the above couplings is given in Section 7 below.

## 2.3. A COUPLING OF MIKE SHE AND MIKE 11

The focus in MIKE SHE lies on catchment processes with a comparatively less advanced description of river processes. In contrary, MIKE 11 has a more advanced description of river processes and a simpler catchment description than MIKE SHE. Hence, for cases where full emphasis is needed for both river and catchment processes a coupling of the two modelling systems is required.

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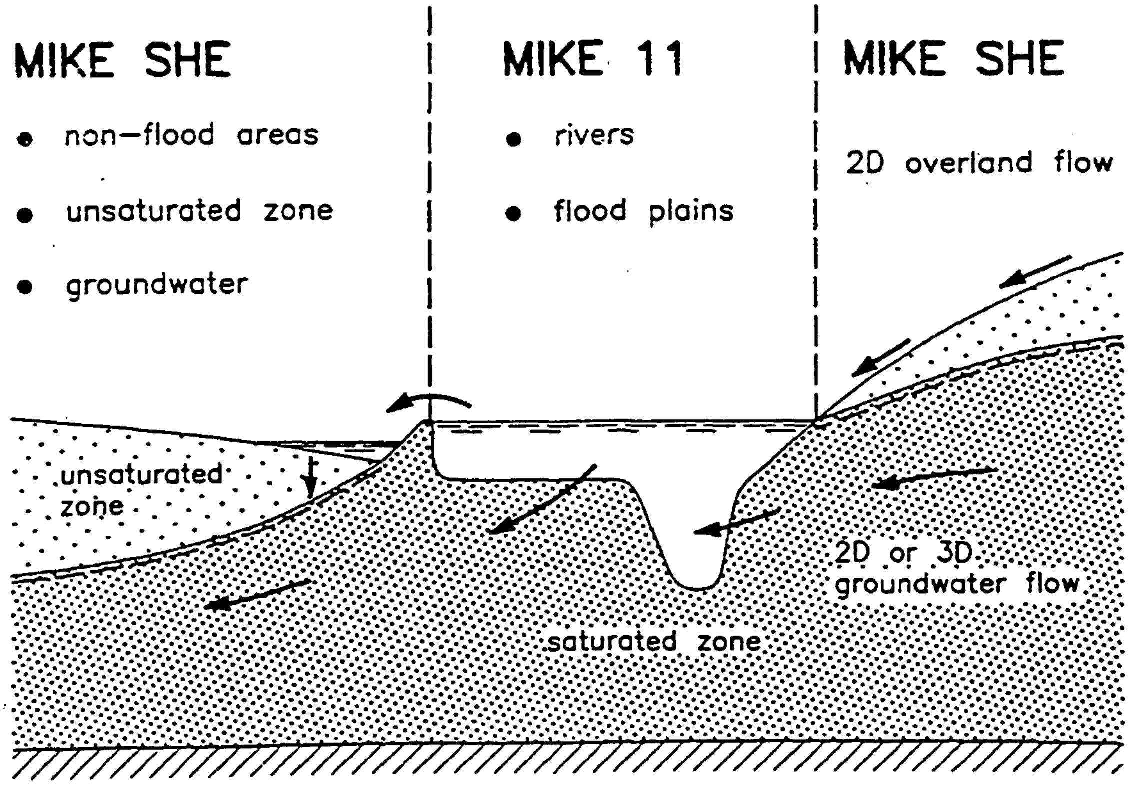


*Figure 4.* Principles of the coupling between the MIKE SHE catchment code and the MIKE 11 river code.

A full coupling between MIKE SHE and MIKE 11 has been developed (Figure 4). In the combined modelling system, the simulation takes place simultaneously in MIKE 11 and MIKE SHE, and data transfer between the two models takes place through shared memory. MIKE 11 calculates water levels in rivers and floodplains. The calculated water levels are transferred to MIKE SHE, where flood depth and areal extent are mapped by comparing the calculated water levels with surface topographic information stored in MIKE SHE. Subsequently, MIKE SHE calculates water fluxes in the remaining part of the hydrological cycle. Exchange of water between MIKE 11 and MIKE SHE may occur due to evaporation from surface water, infiltration, overland flow or river-aquifer exchange. Finally, water fluxes calculated with MIKE SHE are exchanged with MIKE 11 through source/sink terms in the continuity part of the Saint Venant equations in MIKE 11.

The MIKE SHE–MIKE 11 coupling is crucial for a correct description of the dynamics of the river-aquifer interaction. Firstly, the river width is larger than one MIKE SHE grid, in which case the MIKE SHE river-aquifer description is no longer valid. Secondly, the river/reservoir system comprises a large number of hydraulic structures, the operation of which are accurately modelled in MIKE 11, but cannot be accounted for in MIKE SHE. Thirdly, the very complex river branch system with loops and flood cells needs a very efficient hydrodynamic formulation such as in MIKE 11.

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### 2.4. COMPARISON TO OTHER MODELLING SYSTEMS REPORTED IN LITERATURE

Yan and Smith (1994) described the demand and outlined a concept for a full integrated ground water–surface water modelling system including descriptions of hydraulic structures and agricultural irrigation as a decision support tool for water resources management in South Florida. Typical examples of integrated codes described in the literature are Menetti (1995) and Koncsos *et al.* (1995).

In a review of recent advances in understanding the interaction of groundwater and surface water Winter (1995) mainly describes groundwater codes, such as MODFLOW, which have been expanded with some, but very limited, surface water simulation capabilities. The research activities are characterized as '... although studies of these systems have increased in recent years, this effort is minimal compared to what is needed'. Winter (1995) sees the prospects for the future as follows: 'Future studies of the interaction of groundwater and surface water would benefit from, and indeed should emphasise, interdisciplinary approaches. Physical hydrologists, geochemists, and biologists have a great deal to learn from each other, and contribute to each other, from joint studies of the interface between groundwater and surface water.'

Integrated three-dimensional descriptions of flow, transport and geochemical processes is still rarely seen for groundwater modelling of large basins. Thus, according to a recent review of basin-scale hydrogeological modelling (Person *et al.*, 1996) most of the existing reactive transport model codes are based on one-dimensional descriptions.

While many model codes contain a distributed physically-based representation of one of the three main components: ground water, unsaturated zone, and surface water systems, only few codes provide a fully integrated description of all these three main components. For example in an up-to-date book (Singh, 1995) presenting descriptions of 25 hydrological codes only three codes, SHE/SHESED (Bathurst *et al.*, 1995), IHDM (Calver and Wood, 1995) and MIKE SHE (Refsgaard and Storm, 1995) provide such integrated descriptions. Among these three codes only MIKE SHE has capabilities for modelling advection-dispersion and water quality. None of the three codes contained options for computations of hydraulic structures in river systems, nor agricultural modelling such as crop yield and nitrogen leaching.

The individual components of the integrated modelling system presented in this paper, we believe, represent state-of-the-art within their respective disciplines. The uniqueness is the full integration.

## 3. Methodology for Model Construction, Calibration, Validation and Application

The terminology and methodology used in the following is based on the concepts outlined in Refsgaard (1997).

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### 3.1. MODEL CONSTRUCTION

All of the applied models are based on distributed physically-based model codes. This implies that most of the required input data and model parameters can ideally be measured directly in nature.

### 3.2. MODEL CALIBRATION

The calibration of a physically-based model implies that simulation runs are carried out and model results are compared with measured data. The adopted calibration procedure was based on 'trial and error' implying that the model user in between calibration runs made subjective adjustments of parameter values within physically realistic limits. The most important guidance for the model user in this process was graphical display of model results against measured values. It may be argued that such manual procedure adds a degree of subjectivity to the results. However, given the very complex and integrated modelling focusing on a variety of output results and containing a large number of adjustable parameters, automatic parameter optimisation is not yet possible and 'trial and error' still becomes the only feasible method in practise.

### 3.3. MODEL VALIDATION

Good model results during a calibration process cannot automatically ensure that the model can perform equally well for other time periods as well, because the calibration process involves some manipulation of parameter values. Therefore, model validations based on independent data sets are required. To the extent possible, limited by data availability, the models have been validated by demonstrating the ability to reproduce measured data for a period outside the calibration period, using a so-called split-sample test (Klemes, 1986). For some of the models, the model was even calibrated on pre-dam conditions and validated on post-dam conditions, where the flow regime at some locations was significantly altered due to the construction of the reservoir and related hydraulic structures and canals.

### 3.4. MODEL APPLICATION

The validated models have finally been used, as an integrated system, in a scenario approach to assess the environmental impacts of alternative water management options. The uncertainties of the model predictions have been assessed through sensitivity analyses.

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## 4. Selected Results from Model Construction, Calibration and Validation of Individual Components

Comprehensive data collection and processing as well as model calibration and validation were carried out (DHI *et al.*, 1995). In the following sections a few selected results are presented for the individual components. Further aspects of model validation focusing on integrated aspects are discussed in Section 5.

### 4.1. RIVER AND RESERVOIR FLOW MODELLING

The following models have been constructed, calibrated and validated:

- one-dimensional MIKE 11 model for the Danube from Bratislava to Komarno,
- one-dimensional MIKE 11 model for the river branch system at the Slovak floodplain, and
- two-dimensional MIKE 21 model for the reservoir.

The MIKE 11 models have been established in two versions reflecting post- and pre-dam conditions, respectively.

#### 4.1.1. *MIKE 11 River Model for the Danube*

The MIKE 11 model for the Danube is based on river cross-sections measured in 1989 and 1991. The applied boundary conditions were measured daily discharges at Bratislava (upstream) and a discharge rating curve at Komarno (downstream). The model was initially calibrated for two steady state situations reflecting a low flow situation (905 $m^3$ $s^{-1}$) and a flow situation close to the long term average (2390 $m^3$ $s^{-1}$), respectively. Subsequently, the model was calibrated in a nonsteady state against daily water level and discharge measurements from 1991. The model was finally validated by demonstrating the ability to reproduce measured daily water level data from 1990. Calibration and validation results are presented in Topolska and Klucovska (1995). For the post-dam model some river reaches were updated with cross-sections measured in 1993. In addition, the reservoir and related hydraulic structures and canals were included. As the conditions after damming of the Danube have changed significantly, re-calibration of the post-dam model was carried out for the period April 1993–July 1993. Subsequently, the model was validated against measured data from the period November 1992–March 1993.

#### 4.1.2. *MIKE 11 Model for the River Branch System*

The Danubian floodplain is a forest area of major ecological interest characterised by a complex system of river branches. A layout of the river branch system is shown in Figure 5. The cross-sections in the river branch system were measured during the 1960's and 1970's. The pre-dam model was calibrated against water level and flow data from the 1965 flood. In the post-dam situation, the branch system is fed by an

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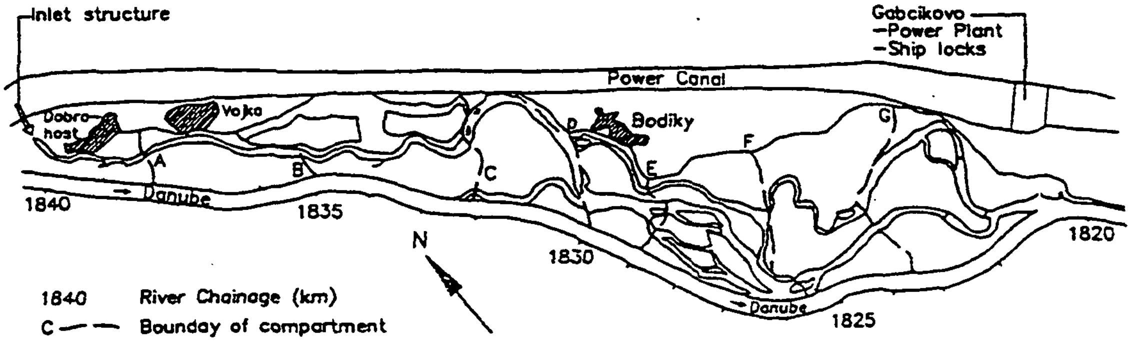


*Figure 5.* Layout of the river branch system on the Slovakian side of the Danube.

inlet structure with water from the power canal. The system consists of a number of compartments (cascades) separated by small dikes. On each of these dikes combined structures of culverts and spillways are located enabling some control of the water levels and flows in the system. Results of the model calibration against data measured during the summer 1994 are shown in Klucovska and Topolska (1995). Finally, the model was validated by demonstrating the ability to reproduce water levels measured during the summer of 1993. Some of these results are presented in Sørensen *et al.* (1996).

### 4.1.3. *MIKE 21 Reservoir Model*

A MIKE 21 hydrodynamic model for the reservoir was established based on a reservoir bathymetry measured in 1994. The spatial resolution of the finite difference model is $100 \times 50$ m. The model was calibrated against flow velocities measured in the reservoir in the autumn of 1994.

## 4.2. GROUND WATER FLOW MODELLING

Ground water modelling has been carried out at three different spatial scales:

- A *regional* ground water model for pre-dam conditions (3000 km$^2$, 500 m horizontal grid, 5 vertical layers).
- A *regional* ground water model for post-dam conditions (3000 km$^2$, 500 m horizontal grid, 5 vertical layers).
- A *local* ground water model for an area surrounding the reservoir for both pre- and post-dam conditions (200 km$^2$, 250 m horizontal grid, 7 vertical layers).
- A *local* ground water model for the river branch system for both pre- and post-dam conditions (50 km$^2$, 100 m horizontal grid, 2 vertical layers).
- A *cross-sectional* (vertical profile) model near Kalinkovo at the left side of the reservoir (2 km long, 10 m horizontal grid, 24 vertical layers).

The regional and local ground water models all use the coupled version of the MIKE SHE and MIKE 11 and hence, include modelling of evapotranspiration and

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snowmelt processes, river flow, unsaturated flow and ground water flow. The cross-sectional model only includes ground water processes.

### 4.2.1. *Model Construction*

Comprehensive input data were available and used in the construction of the models. In general, the regional and the local models are based on the same data with the main difference being that the local models provide finer resolutions and less averaging of measured input data. The two regional models, reflecting pre- and post-dam conditions, are basically the same. The only difference is that the post-dam model includes the reservoir and related hydraulic structures and seepage canals.

The models are based on information on location of river systems and cross-sectional river geometry, surface topography, land use and cropping pattern, soil physical properties and hydrogeology. In addition, time series of daily precipitation, potential evapotranspiration and temperature as well as discharge inflow at Bratislava have been used. Comprehensive geological data exist from this area, see e.g., Mucha (1992) and Mucha (1993). The aquifer, ranging in thickness from about 10 m at Bratislava to about 450 m at Gabcikovo, consists of Danube river sediments (sand and gravel) of late Tertiary and mainly Quaternary age. The present model is based on the work of Mucha *et al.* (1992a, b).

### 4.2.2. *Model Calibration*

The ground water model was calibrated against selected measured time series of ground water levels. The following parameters were subject to calibration: specific yield in the upper aquifer layer, leakage coefficients for the river bed and hydraulic conductivities for the aquifer layers. The soil physical characteristics for the unsaturated zone have been adopted directly from the unsaturated zone/agricultural modelling.

The river model that has been used in the ground water modelling is identical to the MIKE 11 river model of the Danube, which was successfully validated independently as a 'stand alone model' (Subsection 4.1, above). When coupling MIKE SHE and MIKE 11 water is exchanged between the two models. The amount of water that recharges the aquifer in the upstream part and re-enters the river further downstream is in the order of 10–60 $m^3$ $s^{-1}$ depending on the Danube discharge and on the actual ground water level. The recharge is typically two orders of magnitude less than the Danube discharge, and hence, a re-calibration of the MIKE 11 river model is not required. As the major part of the ground water recharge originates from infiltration through the river bed, the leakage coefficient for the river bed becomes very important. Limited field information was available on this parameter, and hence, it was assumed spatially constant and through calibration assessed to be $5 \times 10^{-5}$ $s^{-1}$ for the Danube and Vah rivers and $5 \times 10^{-6}$ $s^{-1}$ for

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the Little Danube. These values are in good agreement with previous modelling experiences (Mucha *et al.*, 1992b).

When keeping the specific yield and the leakage coefficients for the river bed fixed the main calibration parameters were the hydraulic conductivities of the saturated zone. About 300 time series of ground water level observations were available for the model area, typically in terms of 30–40 yr of weekly observations. The calibration was carried out on the basis of about 80 of these series for the period 1986–1990. In the parameter adjustments the overall spatial pattern described in the geological model were maintained. Some of the calibration results are illustrated in Figure 6 showing observed Danube discharge data together with simulated and measured ground water levels for three wells located at different distances from the Danube. Wells 694 and 740 are seen to react relatively quickly to fluctuations in river discharge as compared to well 7221, which is located further away from the river. This illustrates how the dynamics of the Danube propagates and is dampened in the aquifer.

#### 4.2.3. *Model Validation*

The calibrated ground water model was validated by demonstrating the ability to reproduce measured ground water tables after damming of the Danube. In this regard the only model modification is the inclusion of the reservoir and related structures and canals. Due to the nonstationarity of the hydrological regime such a validation test, which according to Klemes (1986) is denoted a differential split-sample test, is a demanding test. Figure 7 shows the simulated and observed ground water levels for the same three observation wells as shown for the calibration period in Figure 6. The effects of the damming of the Danube in October 1992, when the new reservoir was established, is clearly seen in terms of increased ground water levels and reduced ground water dynamics when comparing the two figures. These features are well captured by the model.

### 4.3. GROUND WATER QUALITY

A geochemical field investigation was carried out in a cross-section north of the reservoir near Kalinkovo as a basis for identifying the key geochemical processes and estimating parameter values (see Mucha, 1995). Eleven multi-screen wells were installed close to the water supply wells at Kalinkovo forming a 7.5 km long cross-section parallel to the regional ground water flow direction. The multi-screen wells have been sampled frequently to investigate the ongoing bio-geochemical processes during infiltration of the Danube river water into the aquifer.

A ground water quality model was established for the Kalinkovo cross-sectional profile based on all the measured field data. This model includes a comprehensive description of the bio-geochemical processes such as kinetically controlled denitrification and equilibrium controlled inorganic chemistry based on the well known PHREEQE code. More details are given in Griffioen *et al.* (1995) and

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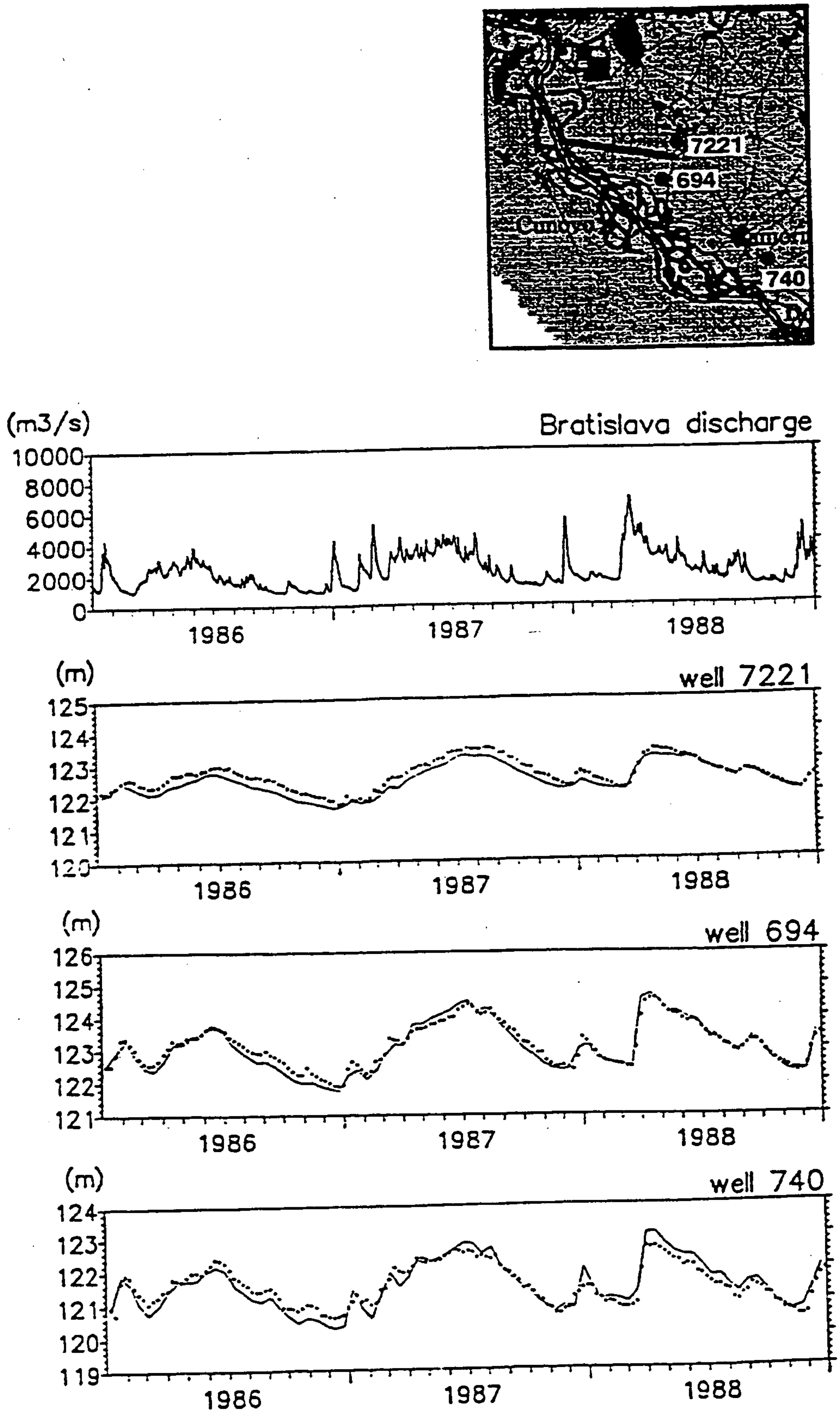


*Figure 6.* Danube discharge at Bratislava together with simulated and observed ground water levels for three wells before the damming of the Danube (calibration period).



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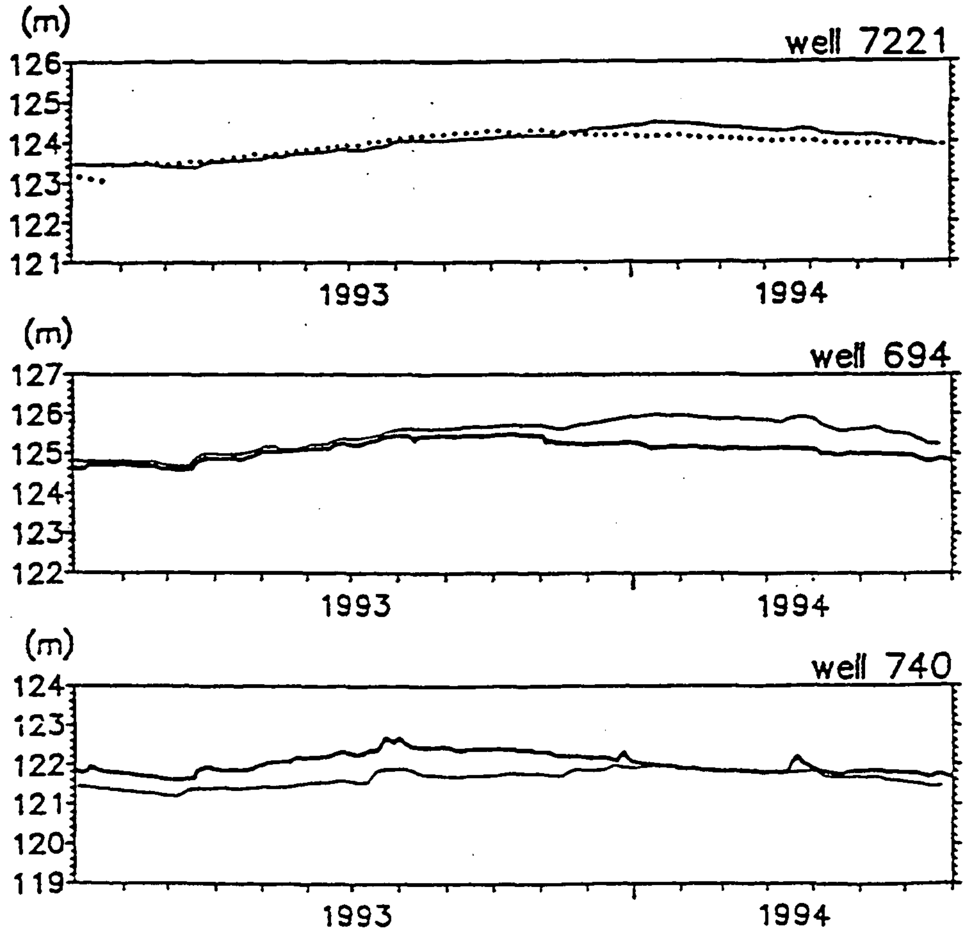


*Figure 7.* Simulated and observed ground water levels for three wells after damming of the Danube (validation period).

Engesgaard (1996). The transport part of the Kalinkovo cross-section has been calibrated against $^{18}O$ isotope data. The parameters describing reactive processes have been assessed and adjusted on the basis of the detailed field measurements in the Kalinkovo cross-sectional profile. It was shown that the geochemical model behaves qualitatively correct (Engesgaard, 1996).

## 4.4. UNSATURATED ZONE AND AGRICULTURAL MODELLING

Modelling of the pre-dam and post-dam conditions of agricultural potential and nitrate leaching risk was carried out using a representative selection of soil units, cropping pattern and meteorological data covering the area between Danube and Maly Danube (Figure 1). The DAISY model uses time-varying ground water levels (simulated with the regional MIKE SHE ground water model) as lower boundary condition, for the unsaturated flow simulations. Cropping pattern and fertiliser application is included in the model based on measurements and statistical data.

The model was calibrated on the basis of data from field experiments carried out during the years 1981–1987 at the experimental station in Most near Bratislava. During this process the crop parameters used in the model were adjusted to Slovak conditions. After the initial model construction and calibration, the model performance was evaluated through preliminary simulations using data from a number of plots located on an experimental field site at Lehnice in the middle of the project area. On the basis of comparisons between measured and simulated values of nitrogen uptake, dry matter yield and nitrate concentrations in soil moisture, the model performance under Slovak conditions was considered satisfactory (DHI *et al.*, 1995).

## 4.5. RIVER AND RESERVOIR SEDIMENT TRANSPORT MODELLING

### 4.5.1. *Danube River Sediment Transport*

A one-dimensional morphological model was established for the Danube. The model operates with cross-sectional averaged parameters representing the river reach between every computational point (i.e. approximately 500 m), a special technique for comparing 'real' and simulated state variables was required. Therefore, the changes in mean water level over a decade rather than changes in bed elevations were compared between observations and simulations. For this purpose the changes in the so-called 'Low Regulation and Navigable Water Level' (LR-NWL) were used. LR-NWL is specified by the Danube Commission as the water level corresponding to $Q_{94\%}$ which is approximately 980 $m^3$ $s^{-1}$. By using such an approach, perturbations in bed levels from one cross-section to another did not destroy the picture of the overall trends in aggradation and degradation of the river bed. The results of the calibration (1974–84) and validation runs (1984–90) are described in Topolska and Klucovska (1995).

### 4.5.2. *Sediment Transport in the River Branch System*

A one-dimensional fine sediment model was constructed for the river branch system in order to have a tool for quantitative evaluation of the possible sedimentation in the river branch system for alternative water management options. The upstream boundary condition for the model was provided in terms of concentration of suspended sediments simulated by the reservoir model. As virtually no field data on sedimentation in the river branch system were available neither calibration nor validation was possible. Instead, experienced values of model parameters from other similar studies as reported in the literature were used.

### 4.5.3. *Reservoir Sediment Model*

A two-dimensional fine graded sediment model was constructed for the reservoir. The suspended sediment input was imposed as a boundary condition in Bratislava with time series of sediment concentrations of six suspended sediment fractions with their own grain sizes and fall velocities. The fall velocity for each of the six fractions was assessed according to field measurements. No further model calibration was carried out. The only field data available for validation were a few bed

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sediment samples from summer 1994 with data on sedimentation thickness and grain size analyses (Holobrada *et al.*, 1994). A comparison of model results and field data indicated that a reservoir sedimentation of the right order of magnitude was simulated. The simulated reservoir sedimentation corresponded to 42% of the total suspended load at Bratislava.

## 4.6. SURFACE WATER QUALITY MODELLING

### 4.6.1. *Danube River Model*

A BOD-DO model (MIKE 11 WQ) has been used to describe the water quality in the main stream of the Danube between Bratislava and Komarno. This model describes oxygen concentration (DO) as a function of the decay of organic matter (BOD), transformation of nitrogen components, re-aeration, oxygen consumption by the bottom and oxygen production and respiration by living organisms. As the conditions from pre-dam to post-dam have changed significantly, separate calibrations and validations were carried out. The pre-dam model was calibrated against data from October 1991 and validated against data from April and August/September 1991. The post-dam model was calibrated against data from May 1993 and validated against data from June 1993.

### 4.6.2. *Model for the River Branch System*

The water quality in the river branches was simulated with a eutrophication model (MIKE 11 EU), in which the algae production is the driving force. The algae growth in this model is described as a function of incoming light, transparency of the water, temperature, sedimentation and growth rate of the algae and of the available inorganic nutrients. The calibration was carried out on the basis of few data available during the period June–August 1993. Due to lack of further data no independent model validation was possible and hence, the uncertainties related to applying the model for making quantitative predictions of the effects of alternative water management schemes may be considerable.

### 4.6.3. *Reservoir Model*

In the reservoir the driving force is also the algae growth and hence, a eutrophication model (MIKE 21 EU) was applied. The reservoir model was calibrated against measured data from August 1994. This field programme was substantial and resulted in much more data than available for the river branch system. Good correspondence between simulated and observed values were achieved during the calibration period. However, no further data have been available for independent validation tests.

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## 5. Validation of Integrated Model

The model calibration and validation have basically been carried out for the individual models using separate domain data for river system, aquifer system, etc. Rigorous validation tests of the integrated model were generally not possible due to lack of specific and simultaneous data on the processes describing the various couplings. Furthermore, although reasonable good assessments of uncertainties of the individual model predictions could be made, it was not obvious how such uncertainty would propagate in the integrated model.

It can be argued that uncertainties in output from one model would in principle influence the uncertainties in other components of the integrated modelling system, thus adding to the total uncertainty of the integrated model. Following this line of argument would lead to the conclusion that the uncertainty of predictions by the integrated model would be larger than the corresponding uncertainty of predictions made by traditional individual models. On the other hand it can also be argued that in the integrated modelling approach the uncertainties in the crucial boundary conditions are reduced, because assumptions needed for executing individual models are substituted by model simulations based on data from neighbouring domains, which, if properly calibrated and validated, better represent the boundary effects. This would lead to the conclusion that the uncertainties in predictions by the integrated model would be smaller than those of the individual models.

In the present study, no theoretical analyses have been made of this problem. Instead, a few validation tests have been made for cases where the couplings could indirectly be checked by testing the performance of the integrated model against independent data. In the following, results from one of these validation tests for the integrated model are shown.

The river-aquifer interaction changed significantly, when the reservoir was established. An important model parameter describing this interaction is the leakage coefficient, which was calibrated on the basis of ground water level data for the pre-dam situation (Subsection 4.2). For the post-dam situation the MIKE 21 reservoir model calculates the thickness and grain sizes of the sedimentation at all points in the reservoir. By use of the Carman-Kozeny formula, the leakage factors are recalculated for the area which was now covered by the reservoir. The model results were then checked against ground water level observations from wells near the reservoir, and it was found, that a calibration factor of 10 had to be applied to the Carman-Kozeny formula. This can theoretically be justified by the fact that the sediments are stratified or layered due to variations in flow velocities during the sedimentation process. The same formula and the same calibration factor was also used for converting all texture data from aquifer sediment samples to hydraulic conductivity values in the model.

Now, how can the validity of the integrated model be tested ? The ground water level observations from a few wells have been used to assess the leakage calibration factor, so although the model output was subsequently checked against data from

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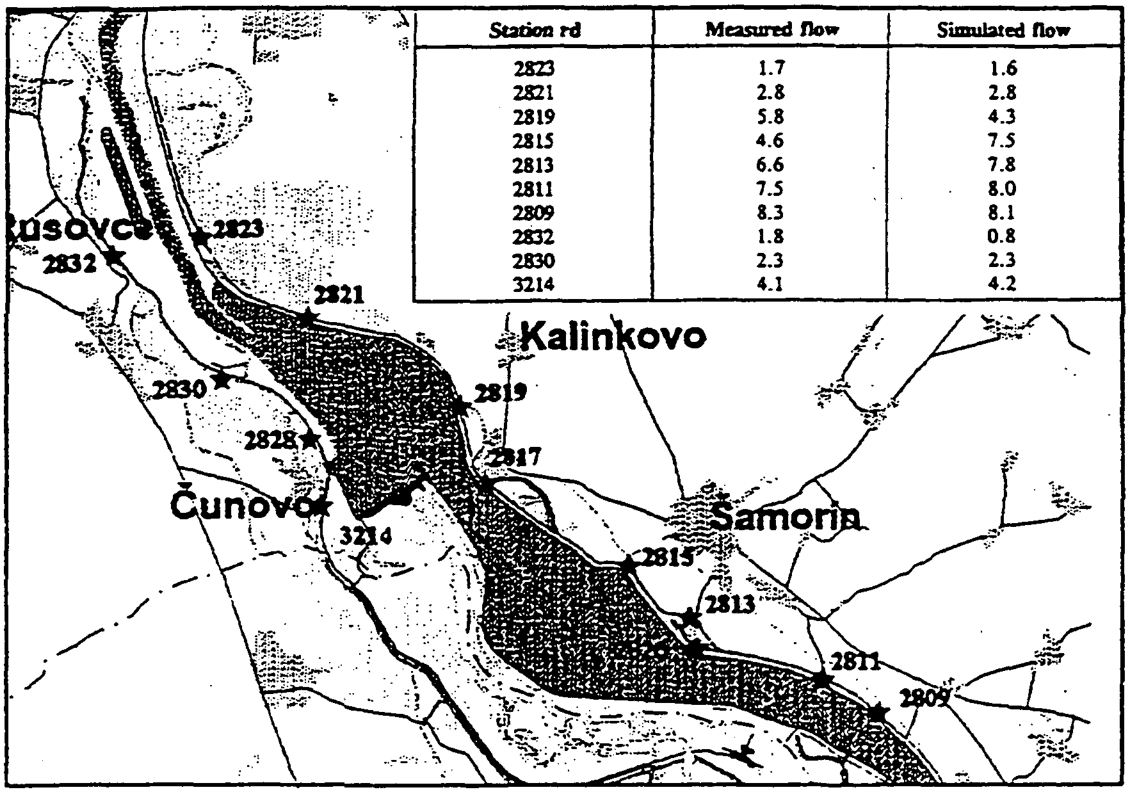


| Station rd | Measured flow | Simulated flow |
|---|---|---|
| 2823 | 1.7 | 1.6 |
| 2821 | 2.8 | 2.8 |
| 2819 | 5.8 | 4.3 |
| 2815 | 4.6 | 7.5 |
| 2813 | 6.6 | 7.8 |
| 2811 | 7.5 | 8.0 |
| 2809 | 8.3 | 8.1 |
| 2832 | 1.8 | 0.8 |
| 2830 | 2.3 | 2.3 |
| 3214 | 4.1 | 4.2 |

*Figure 8.* Measured and simulated discharges in seepage canals. The data are from a particular day in May 1995 and in $m^3 s^{-1}$.

many more wells, it may be argued that this in itself is not sufficient for a true model validation. Consider instead a comparison of simulated and measured discharges in the so-called seepage canals, which are small canals constructed a few hundred meters away from the reservoir with the aim of intercepting part of the infiltration through the bottom of the reservoir. In Figure 8 it can be seen that the model simulations match the measured data remarkably well at different locations along the seepage canals. Thus, at the two stations most downstream on both seepage canals (stations 2809 and 3214) the agreements between model predictions and field data are within 5%. This is a powerful test, because the discharge data have not been used at all in the calibration process, and because it integrates the effects of reservoir sedimentation, calculation of leakage factors and geological parameters.

## 6. Model Application – Case Study of River Branch System

### 6.1. HYDROLOGY OF RIVER BRANCH SYSTEM

The hydrology of the river branch system is highly complex with many processes influencing the water characteristics of importance for flora and fauna (Figure 2). These processes are highly interrelated and dynamic with large variations in time and space. The complexity of the floodplain, with its river branch system, is indicated in Figures 5 and 9 for the 20 km reach downstream the reservoir on the Slovakian side, where alluvial forest occurs. Before the damming of the Danube

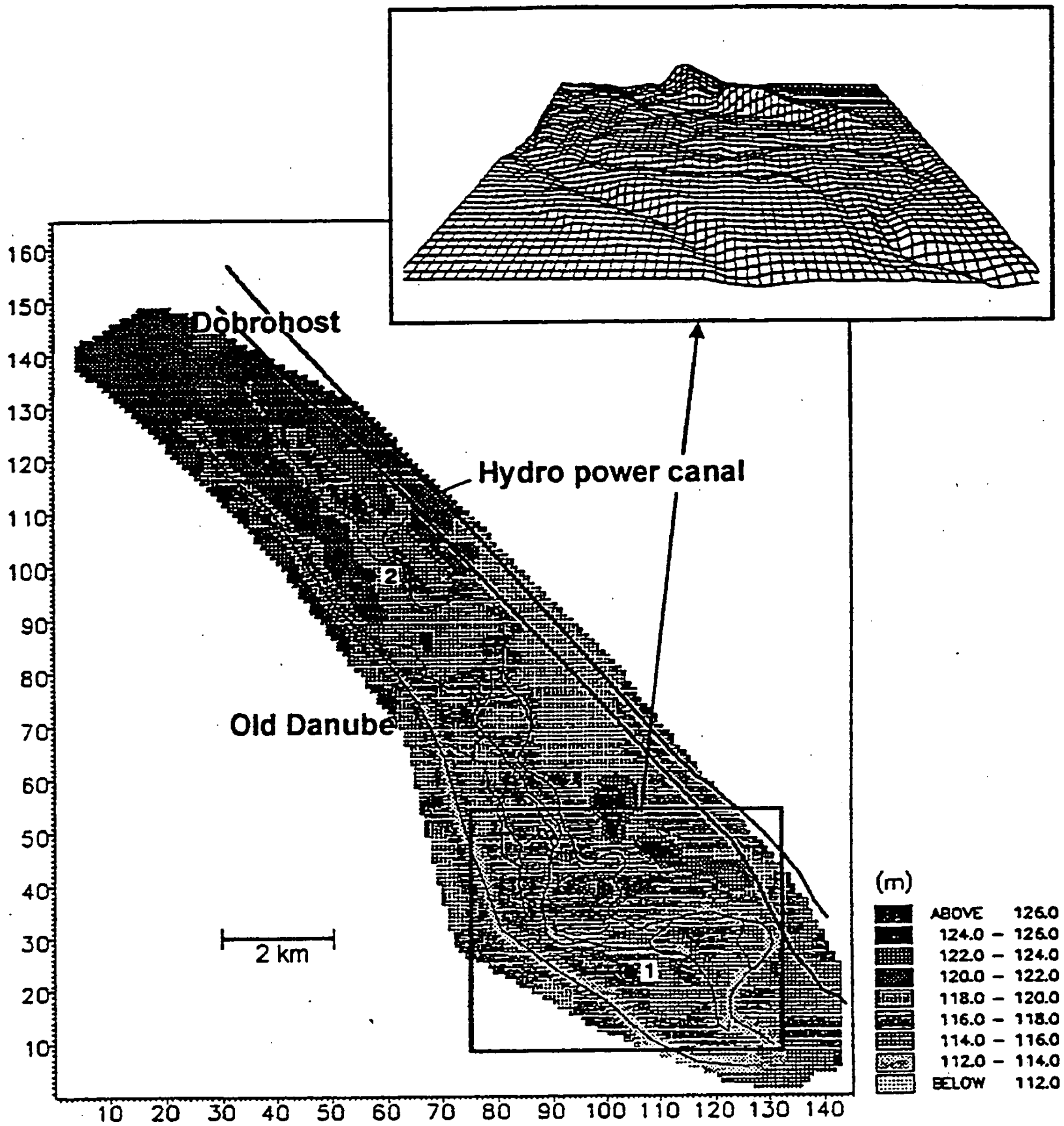


*Figure 9.* Plan and perspective view of the surface topography, of the river branches and the related flood plains as represented in a model network of 100 m grid squares.

in 1992 the river branches were connected with the Danube during periods with discharge above average. However, some of the branches were only active during flood situations a few days per year. It was anticipated that after the damming, the water level in the Danube would decrease significantly. Therefore, in order to avoid that water drains from the river branches to the Danube, resulting in totally dry river branches, the water outflow from branches into the Danube have been blocked except for the downstream one at chainage 1820 rkm (Figure 5). Now, the river branch system receives water from an inlet structure in the hydropower canal at Dobrohost (Figure 5). This weir has a design capacity of 234 $m^3 s^{-1}$. Together with the various hydraulic structures in the river branches, it controls the hydraulic, hydrological and ecological regime in the river branches and on the flood plains.

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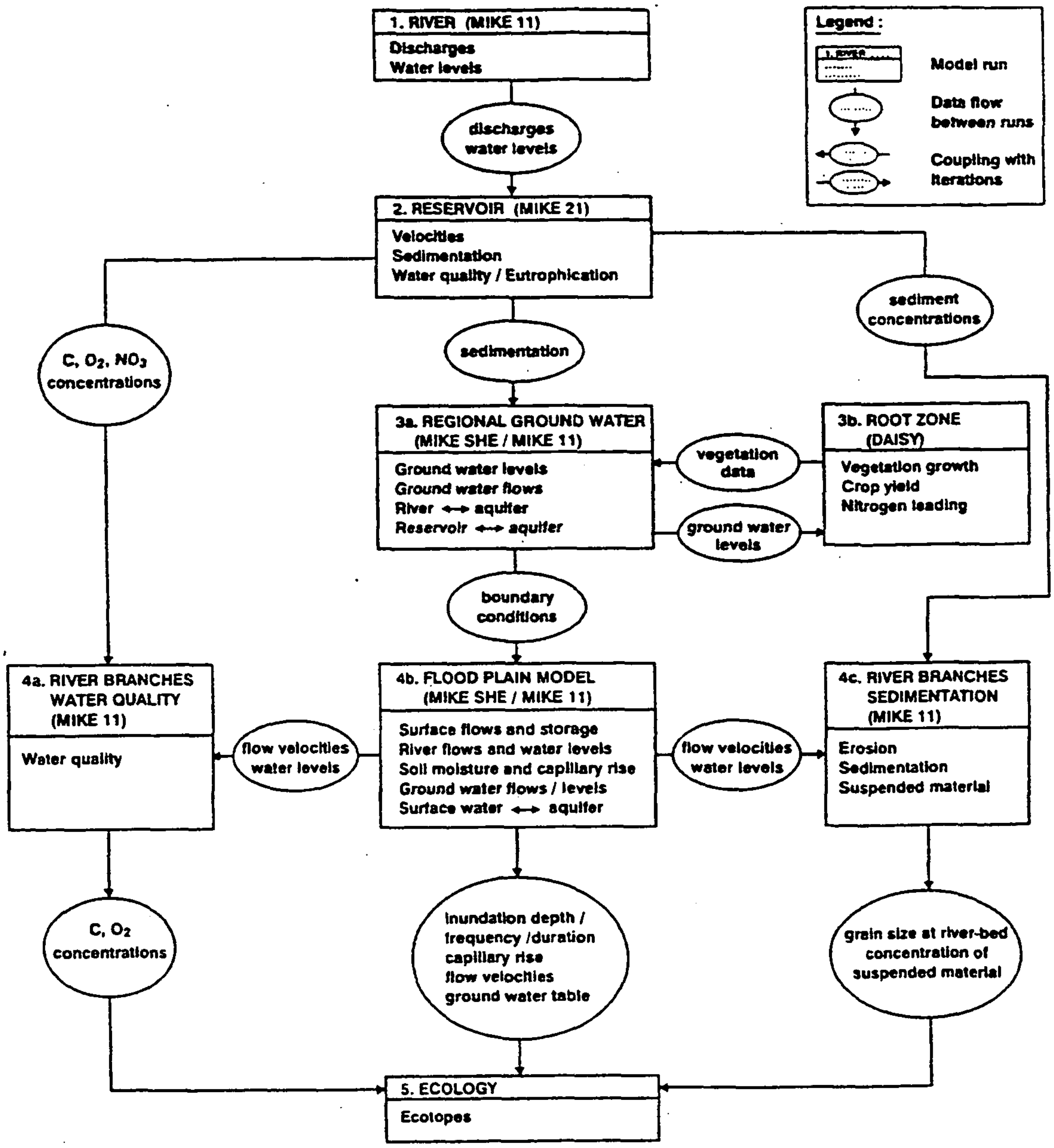


*Figure 10.* Steps in integrated model for floodplain hydrology.

## 6.2. MODELLING APPROACH

Comprehensive field studies and modelling analyses are often carried out in connection with assessing environmental impacts of hydropower schemes. Recent examples from the Danube include the studies of the Austrian schemes Altenwörth (Nachtnebel, 1989) and Freudenau (Perspektiven, 1989). However, like in the Austrian cases, the modelling studies have most often been limited to independent modelling of river systems, groundwater systems or other subsystems, without providing an integrated approach as the one presented in this paper.

The models in this study were applied in a scenario approach simulating the hydrological conditions resulting from alternative possible operations of the entire system of hydraulic structures (alternative water management regimes). Thus, one historical (pre-dam) regime and three hypothetical (post-dam) water regimes cor-

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responding to alternative operation schemes for the structures of the Gabcikovo system were simulated (DHI *et al.*, 1995). Due to the integration of the overall modelling system each scenario simulation involves a sequence, some times in an iterative mode, of model calculations. For the case of river branch modelling a hierarchical scheme of simulation runs (Figure 10) included the following major steps:

*Step 1. Hydraulic river modelling (MIKE 11)*

*Model simulation*: The MIKE 11 model simulates the river flows and water levels in the entire river system and river branches.

*Coupling*: The model outputs, in terms of flows into the reservoir at the upstream end and downstream outflows through the reservoir structures are used as boundary conditions for the reservoir modelling (Step 2). Furthermore, the flow velocities and water levels are used in the river water quality simulations (Step 4a).

*Step 2. Reservoir modelling (MIKE 21)*

*Model simulation:* The MIKE 21 reservoir model simulates velocities, sedimentation and eutrophication/water quality in the reservoir.

*Coupling*: The flow boundary conditions are generated by the river model (Step 1). Results on sedimentation are used to calculate leakage coefficients. Results on oxygen, nitrogen and carbon can be used as boundary conditions of river water quality, water quality of infiltrating water (Step 3a).

*Step 3a. Regional ground water flow (MIKE SHE/MIKE 11)*

*Model simulation:* The coupled MIKE SHE/MIKE 11 model simulates the ground water flow and levels including the interaction with the river system and the reservoir.

*Coupling:* In the reservoir, the infiltration is simulated on the basis of leakage coefficients, which have been calculated from the amount and composition (grain sizes) of the sedimentation on the reservoir bottom (Step 2). This link between reservoir sedimentation and ground water was shown to be crucial for the model results. Furthermore, an iterative link to the DAISY agricultural model exists (Step 3b). Hence, spatially and temporally varying ground water levels from MIKE SHE/MIKE 11 are used as lower boundary conditions in DAISY, which in turn simulates the leaf area index and the root zone depth which are used as input time series data in MIKE SHE/MIKE 11. The model outputs, in terms of ground water flow velocities, are used as input to the ground water quality simulation. The model results, in terms of river flow velocities and water levels, ground water flow velocities and water levels, are used as time varying boundary conditions for the local flood plain model (Step 4b).

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*Step 3b. Root zone (DAISY)*

*Model simulation*: The DAISY model simulates the unsaturated zone flows, the vegetation development, including crop yield.

*Coupling*: The DAISY has an iterative link to the MIKE SHE/MIKE 11 model (as described above under Step 3a).

*Step 4a. River branches water quality (MIKE 11)*

*Model simulation*: The MIKE 11 model simulates the river water quality (BOD, DO, COD, $NO_3$, etc).

*Coupling*: The model uses data from Step 2 and Step 4b and produces output on concentrations of COD and DO, which are used as input to the ecological assessments (Step 5).

*Step 4b. Flood plain model (MIKE SHE/MIKE 11)*

*Model simulation*: The coupled MIKE SHE/MIKE 11 model simulates all the flow processes in the flood plain area including water flows and storages on the ground surface, river flows and water levels, ground water flows and water levels, evapotranspiration, soil moisture content in the unsaturated zone and capillary rise.

*Coupling*: The model uses data from Step 3a as boundary conditions and provides river flow velocities as the basis for the water quality and sediment simulations (Steps 4a and c). The model provides data on flood frequency and duration, depth of flooding, depth to ground water table, moisture content in the unsaturated zone and flow velocities in river branches, which are key figures in the subsequent ecological assessments (Step 5).

*Step 4c. River branches sedimentation (MIKE 11)*

*Model simulation*: The MIKE 11 model simulates the transport of fine sediments through the river branch system. As a result the sedimentation/erosion and the suspended sediment concentrations are simulated.

*Coupling*: The model uses sediment concentrations simulated by the reservoir model (Step 2) as input. Furthermore, the flow velocities simulated by the local flood plain model (Step 4b) are used as the basis for the sediment calculations. The results, in terms of grain size of the river bed and concentrations of suspended material, are used as input to the ecological assessments (Step 5).

*Step 5. Ecology*

A correlation matrix between the physical/chemical parameters provided by the model simulations (Steps 4a, b and c) and the aquatic and terrestric ecotopes has been established for the project area. Alternative water management regimes can be described in terms of specific operation of certain hydraulic structures and corresponding distribution of water discharges primarily between the Danube, the Gabcikovo hydropower scheme and the river branch

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system. The hydrological effects of such alternative operations can be simulated by the integrated model and subsequently, the ecological impacts can be assessed in terms of likely changes of ecotopes.

### 6.3. THE FLOODPLAIN MODEL

The extent of the floodplain model area is indicated in Figure 5 and a perspective view of the area with the river branch system and floodplains is shown in Figure 9. The horizontal discretization of the finite difference model is 100 m, and the ground water zone is represented by two layers. Several hundreds of cross-sections and more than 50 hydraulic structures in the river branch system were included in the MIKE 11 model for the river system.

For the pre-dam model, the surface water boundary conditions comprise a discharge time series at Bratislava and a discharge rating curve at the downstream end (Komarno). For the post-dam model, the Bratislava discharge time series has been divided into three discharge boundary conditions, namely at Dobrohost (intake from hydropower canal to river branch system), at the inlet to the hydropower canal and at the inlet to Danube from the reservoir. For the groundwater system, time varying ground water levels simulated with the regional ground water models act as boundary conditions. The Danube river forms an important natural boundary for the area. The Danube is included in the model, located on the model boundary, and symmetric ground water flow is assumed below the river. Hence, a zero-flux boundary condition is used for ground water flow below the river.

To illustrate the complex hydrology and in particular the interaction between the surface and subsurface processes model results from a model simulation for a period in June–July 1993 are shown in Figures 11 and 12.

Figure 11 presents the inlet discharges at the upstream point of the river branch system (Dobrohost), while the discharges and water levels at the confluence between the Danube and the hydropower outlet canal downstream of Gabcikovo during the same period are shown in Figure 12. Figure 11 further shows the soil moisture conditions for the upper two m below terrain and the water depth on the surface at location 2. Similar information is shown for location 1 in Figure 12. A soil water content above 0.40 (40 vol.%) corresponds to saturation. Location 2 is situated in the upstream part of the river branch system, while location 1 is located in the downstream part (see Figure 9).

At location 2 (Figure 11) flooding is seen to occur as a result of river spilling (surface inundation occurs *before* the ground water table rises to the surface) whenever the inlet discharge exceeds approximately 60 $m^3\ s^{-1}$. The soil moisture content is seen to react relatively fast to the flooding and the soil column becomes saturated. In contrary, full saturation and inundation does not occur in connection with the flood in the Danube in July, but the event is recognised through increasing ground water levels following the temporal pattern of the Danube flood.

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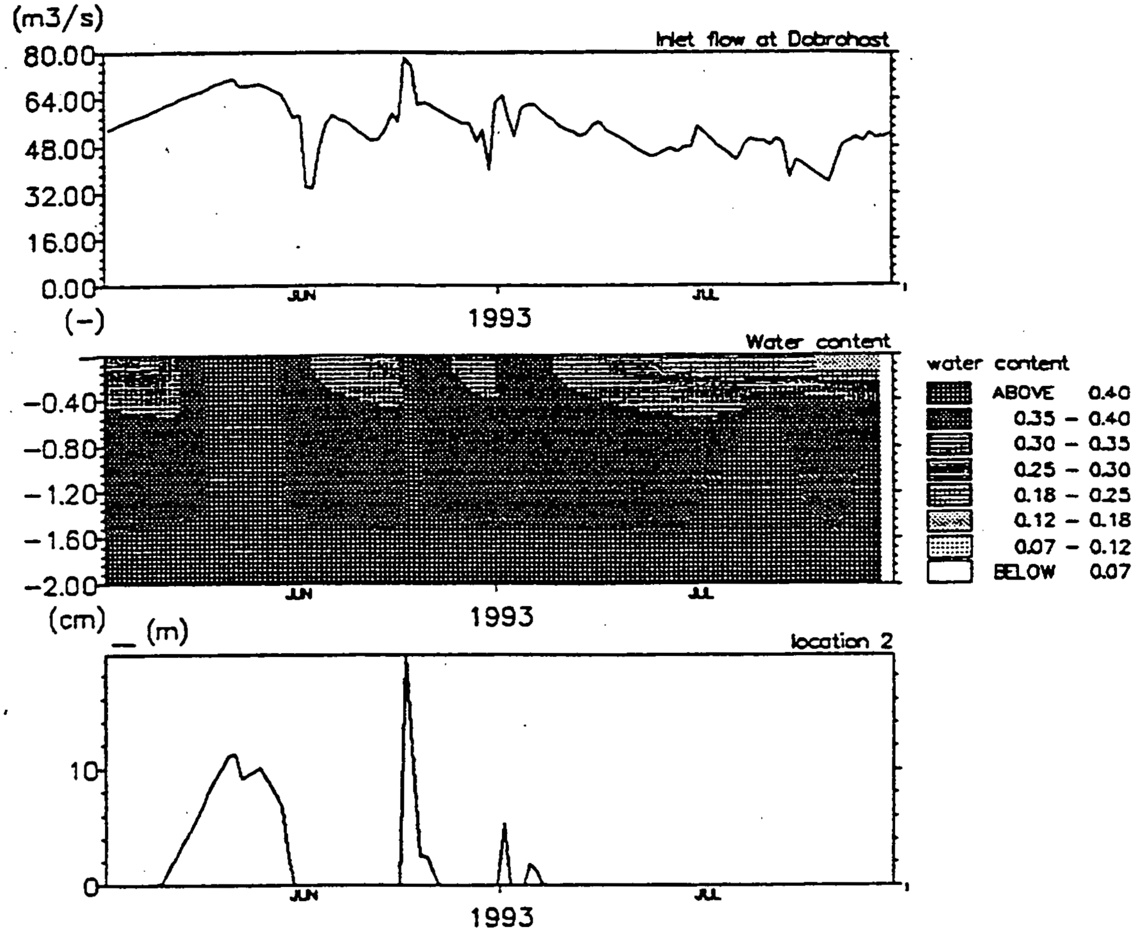


*Figure 11.* Observed inlet discharge to the river branch system at Dobrohost; simulated moisture contents at the upper two m of the soil profile at location 2 and simulated depths of inundation at location 2 during June–July 1993.

At location 1 (Figure 12) the conditions are somewhat different. During the simulation period location 1 never becomes inundated due to high inlet flows at Dobrohost. However, during the July flood in Danube, inundation at location 1 occurs as a result of increased ground water table caused by higher water levels in river branches due to backwater effects from the Danube. The surface elevation at location 1 is 116.4 m which is 0.4 m below the flood water level shown in Figure 12 at the confluence (5 km downstream of location 1). It is noticed that the inundation at this location occurs as a result of ground water table rise and not due to spilling of the river (surface inundation occurs *after* the ground water table has reached ground surface).

### 6.4. EXAMPLE OF MODEL RESULTS

As an example of the results which can be obtained by the floodplain model, Figure 13 shows a characterisation of the area according to flooding and depths to groundwater. The map has been processed on the basis of simulations for 1988 for pre-dam conditions. The classes with different ground water depths and flooding

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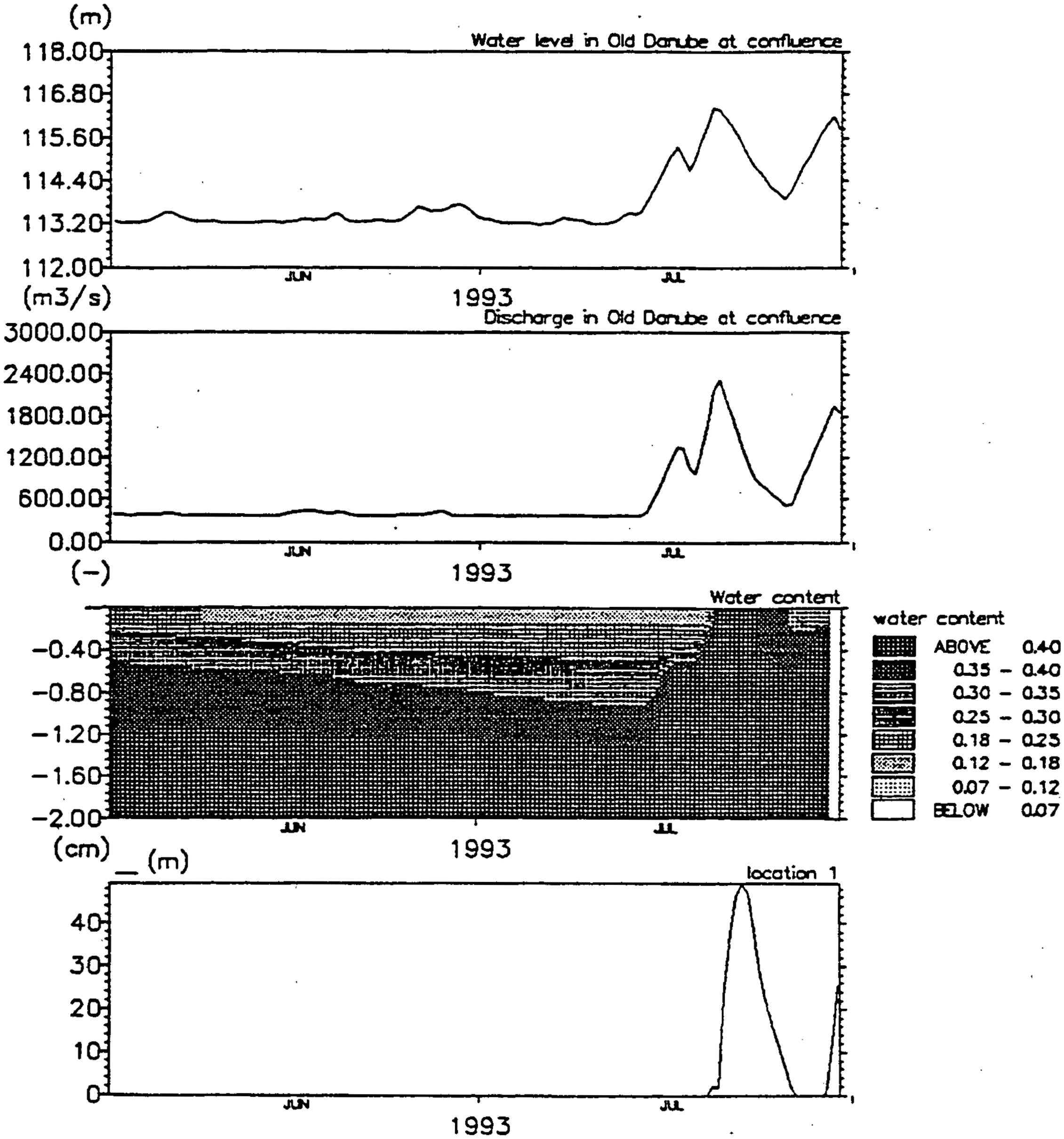


*Figure 12.* Simulated discharge and water levels in the Danube at the confluence between Danube and the outlet canal from the hydropower plant; simulated moisture contents at the upper two meter of the soil profile at location 1 and simulated depths of inundation at location 1 in the river branch system during June–July 1993.

have been determined from ecological considerations according to requirements of (semi)terrestrial (floodplain) ecotopes. From the figure the contacts between the main Danube river and the river branch system is clearly seen. Similar computations have been made by alternative water management schemes after damming of the Danube. The results of one of the hypothetical post-dam water management regimes, characterized by average water flows in the power canal, Danube and river branch system intake of 1470 $m^3 s^{-1}$, 400 $m^3 s^{-1}$ and 45 $m^3 s^{-1}$, respectively, are shown in Figure 14. By comparing Figure 13 and Figure 14 the differences in hydrological conditions can clearly be seen. For instance the pre-dam conditions (Figure 13) are in many places characterised by high groundwater tables

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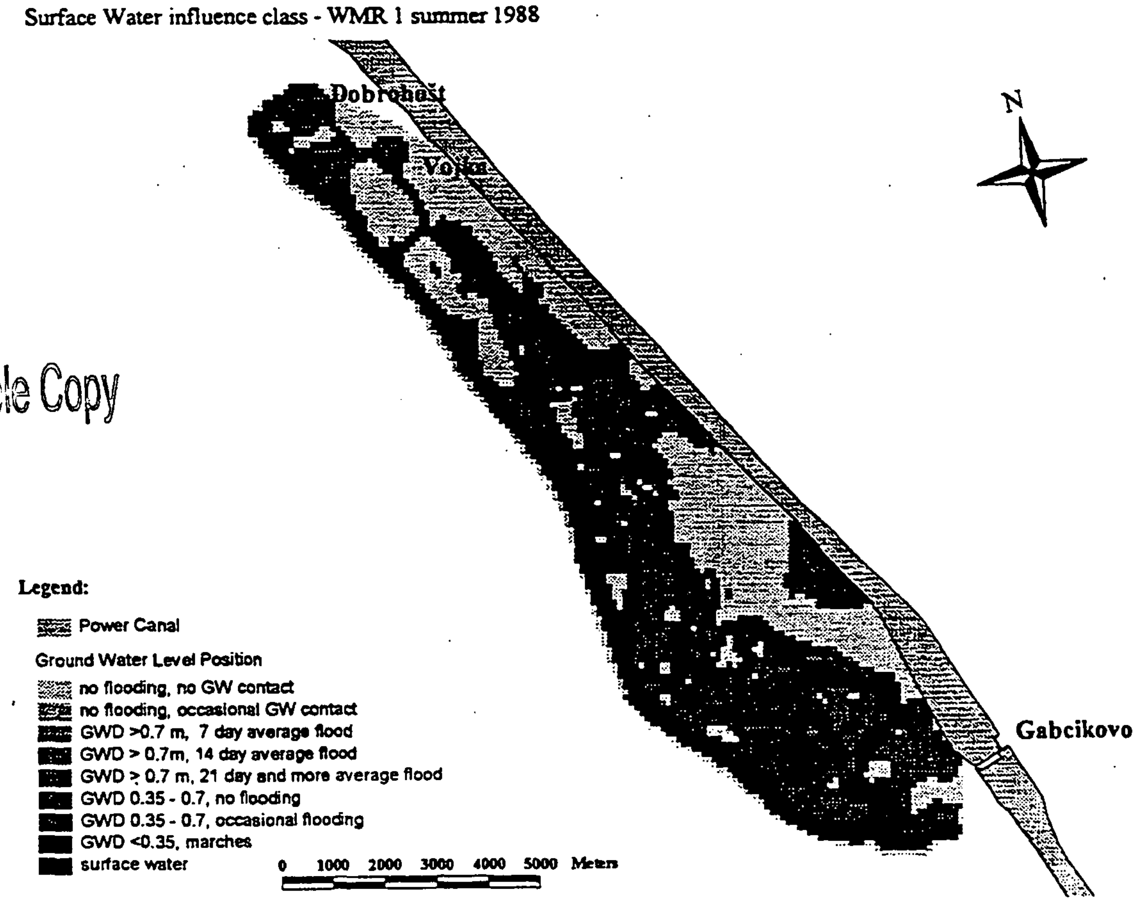




*Figure 13.* Hydrological regime in the river branch area for 1988 pre-dam conditions characterized in ecological classes.

and small/seldom flooding, while the post-dam situation (Figure 14) generally has deeper ground water tables and more frequent flooding. From such changes in hydrological conditions inferences can be made on possible changes in the floodplain ecosystem.

Further scenarios (not shown here) have, amongst others, investigated the effects of establishing underwater weirs in the Danube and in this way improvement of the connectivity between the Danube and the river branch system.

## 7. Limitations in the Couplings made in the Integrated Model

The integrated modelling system and the way it was applied includes different degrees of integration ranging from sequential runs, where results from one model are used as input to the next model, to a full integration, such as the coupling between MIKE SHE and MIKE 11. Hence, the system is not truly integrated in all respects. The justification for these different levels lies in assessments of where it was required in the present project area to account for feed back mechanisms and where such feed backs could be considered to be of minor importance for all practical purposes. For other areas with different hydrological characteristics, the required levels of integration are not necessarily the same. Therefore, a discussion

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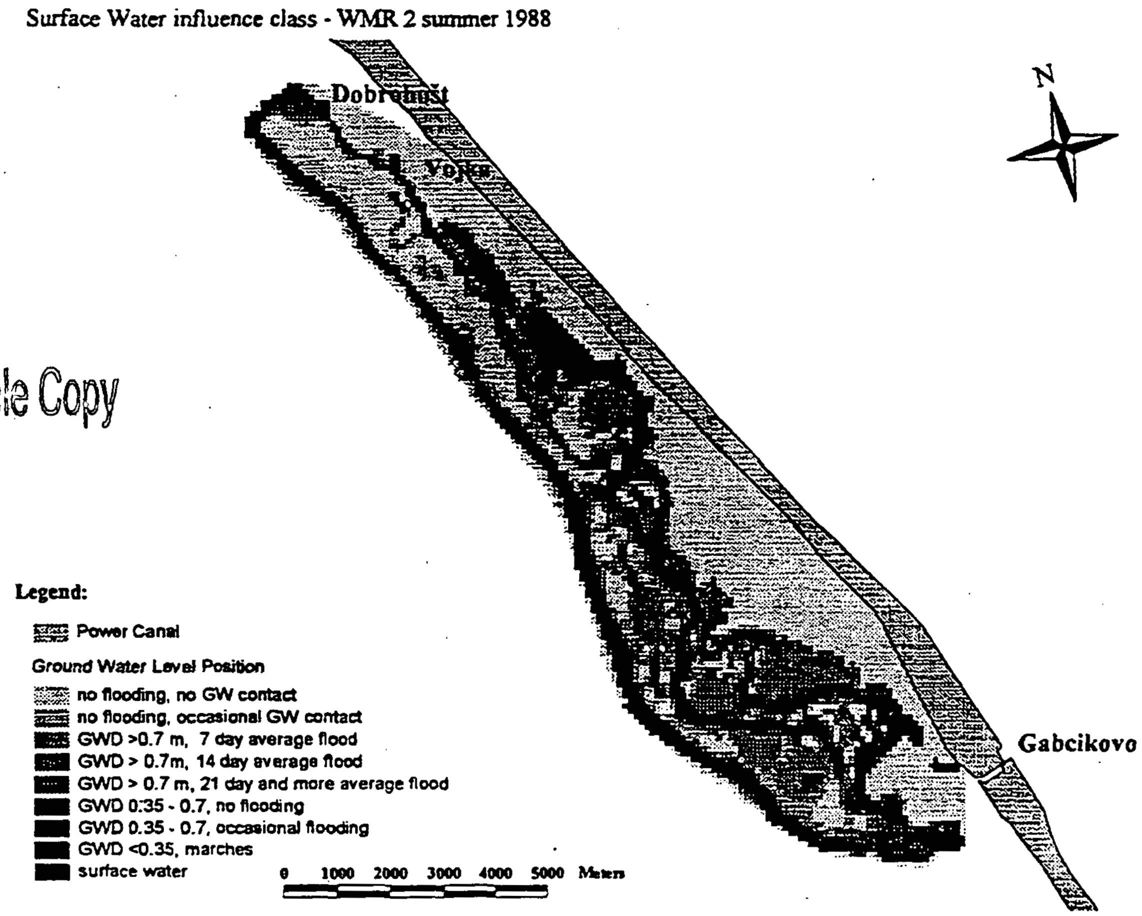




*Figure 14.* Hydrological regime in the river branch area for a post-dam water management regime characterized in ecological classes. The scenario has been simulated using 1988 observed upstream discharge data and a given hypothetical operation of the hydraulic structures.

is given below on the universality and limitations of the various couplings made in the present case.

*A. Hydrological catchment/river hydraulics (MIKE SHE/MIKE 11)*
This coupling between the hydrological code and the river hydraulic code is fully dynamic and fully integrated with feed back mechanisms between the two codes within the same computational time step. This coupling cannot be treated sequentially in this area, since the feedback between river and aquifer works in both directions, with the river functioning as a source in part of the area and as a drain in other parts, and since the direction of the stream-aquifer interaction changes dynamically in time and space as a consequence of discharge fluctuations in the Danube. This coupling was shown to be crucial during the course of the project, and, due to the full integration, it is fully generic.

*B. Reservoir/river (MIKE 21/MIKE 11)*
This coupling is a simple one-way coupling with the reservoir model providing input data to the downstream river model, both in terms of sediment and water

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quality parameters. This coupling is sufficient in the present case, because there is no feedback from the downstream river to the reservoir. Even though this coupling is not fully generic, it may be sufficient in most cases, even in cases with a network of reservoirs and connecting river reaches.

*C. Reservoir/groundwater water exchange (MIKE 21/MIKE SHE)*
This coupling is a simple one-way coupling with the reservoir model providing data on sedimentation to the groundwater module of MIKE SHE, where they are used to calculate leakage coefficients in the surface water/ground water flow calculations. This coupling is sufficient in the present case, where the reservoir water table always is higher than the ground water table, and where the flow always is from the reservoir to the aquifer. However, for cases where water flows in both directions, or where there are significant temporal variations in the sedimentation, the present coupling is not necessarily sufficient.

*D. Hydrology catchment/crop growth (MIKE SHE/DAISY)*
This coupling is an iterative coupling with data flowing in both directions. However, it is not a full integration with the two model codes running simultaneously. Therefore, a number of iterations are required until the input data used in MIKE SHE (vegetation data simulated by DAISY) generates the input data used in DAISY (ground water levels) and vice versa. For example, changes in river water levels affect the ground water levels, implying that the crop growth conditions change and hence, the DAISY simulated vegetation data used by MIKE SHE to simulate the ground water levels are not correct. In such a case, the MIKE SHE simulation has to be repeated with the new crop growth data and subsequently, the DAISY simulation has to be repeated with the new ground water levels, etc., until the differences become negligible. This coupling has been used successfully in previous studies (Styczen and Storm, 1993), but may, due to the iterative mode, be troublesome in practise.

*E. Surface water/ground water quality (MIKE 11 – MIKE 21/MIKE SHE)*
In contrary to the full coupling of flows (coupling A) the corresponding water quality coupling is a simple one-way coupling with the river and reservoir models providing the water quality parameters in the infiltrating water and uses these as boundary conditions for the ground water quality simulations. This coupling is sufficient in the present case with respect to the reservoir, where the flow always is from the reservoir to the aquifer. The river-aquifer interaction involves flows in both directions, but the return flow from the aquifer to the Danube is very small (about 1%) as compared to the Danube flow, and hence, the feedback from the ground water quality to Danube water quality is assumed negligible. However, for other cases where the mass flux from the aquifer to the river system is important for the river water quality, the present one-way coupling will not be sufficient.

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## 8. Discussion and Conclusions

The hydrological and ecological system of the Danubian Lowland is so complex with so many interactions between the surface and the subsurface water regimes and between physical, chemical and biological changes, that an integrated numerical modelling system of the distributed physically-based type is required in order to provide quantitative assessments of environmental impacts on the ground water, the surface water and the floodplain ecosystem of alternative management options for the Gabcikovo hydropower scheme.

Such an integrated modelling system has been developed, and an integrated model has been constructed, calibrated and, to the extent possible, validated for the 3000 km$^2$ area. The individual components of the modelling system represent state-of-the-art techniques within their respective disciplines. The uniqueness is the full integration. The integrated system enables a quite detailed level of modelling, including quantitative predictions of the surface and ground water regimes in the floodplain area, ground water levels and dynamics, ground water quality, crop yield and nitrogen leaching from agricultural land, sedimentation and erosion in rivers and reservoirs, surface water quality as well as frequency, magnitude and duration of inundations in floodplain areas. The computations were carried out on Hewlett Packard Apollo 9000/735 UNIX workstations with 132 MB RAM. With a 300 MHz Pentium II NT computer a typical computational times for one of the steps described in Section 6.2 (Figure 10) would be 2–10 hr. Thus, although the integrated system is rather computationally demanding, the computational requirements are not a serious constraint in practise as compared to the demand for comprehensive field data.

For most of the individual model components, traditional split-sample validation tests have been carried out, thus documenting the predictive capabilities of these models. However, this was not possible for some aspects of the integrated model. Hence, according to rigorous scientific modelling protocols, the integrated model can be argued to have a rather limited predictive capability associated with large uncertainties. A theoretical analysis of error propagation in such an integrated model would be quite interesting, but was outside the scope of the present study which was limited to the comprehensive task of developing the integrated modelling system and establishing the integrated model on the basis of all available data. However, on the basis of the few possible tests (e.g. Figure 7) of the integrated model against independent data not used in the calibration-validation process for the individual models, it is our opinion that the uncertainties of the integrated model are significantly smaller than those of the individual models. The two key reasons for this are: (1) in the integrated model the internal boundaries are simulated by neighbouring model components and not just assessed through qualified but subjective estimates by the modeller; and (2) the integrated model makes it possible to explicitly include more sources of data in validation tests that can not all be utilised in the individual models. Thus, by adding independent validation tests for

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the integrated model, such as the one shown in Figure 7 on discharges in seepage canals, to the validation tests for the individual models, the outputs of the integrated model have been subject to a more comprehensive test based on more data and hence, must be considered less uncertain than outputs from the individual models.

The environmental impacts of the new reservoir and the diversion of water from the Danube through the Gabcikovo power plant can be simulated in rather fine detail by the integrated model established for the area. The integrated nature of the model has been illustrated by a case study focusing on hydrology and ecology in the wetland comprising the river branch system. The integrated model is not claimed to be capable of predicting detailed ecological changes at the species level. However, it is believed to be capable of simulating changes in the hydrological regime resulting from alternative water management decisions to such a degree of detail that it becomes a valuable tool for broader assessments of possible ecological changes in the area.

## Acknowledgements

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# Indvindingsoplande og særlige drikkevandsområder

**Debatten omkring beskyttelsen af grundvandet pågår med uformindsket styrke. Begreber som kildepladszone, indvindingsoplande, infiltrationsområder, beskyttelseszoner og særlige drikkevandsområder svirrer i luften. ... er lagt op til anvendelse af endog betydelige ressourcer i forbindelse med fx tvungne ændringer af arealanvendelse og i forbindelse med udlægning af beskyttelseszoner for grundvandsindvindinger. Der må derfor også ofres ressourcer på at bestemme og prioritere disse områder bedst muligt.**

ANDERS REFSGAARD
JAN GREGERSEN
MICHAEL BUTTS
MICHAEL KRISTENSEN

Udlægning af områder, hvor der pålægges visse restriktioner på arealanvendelsen, fx mindsket brug af pesticider og gødning, for at sikre den fremtidige kvalitet på det nedsivende grund-...d, vil formentlig blive mere og mere ...indeligt. Man er allerede startet i Nordjyllands Amt med egentlig opkøb af landbrugsarealer og efterfølgende udlægning til skovbrug. Der vil derfor blive brug for troværdige metoder til udpegning af de områder, hvor grundvandet egentlig dannes, og formentlig også for at vide, hvor lang tid der går fra grundvandet dannes, til det genfindes i indvindingsboringerne.

I forbindelse med gennemførelsen af Det Strategiske Miljøforskningsprogram, projekt 1.2 og 2, blev der foretaget bestemmelser af grundvandets alder i forskellige boringer beliggende i forskellige dybder og områder, se /2/. Disse målinger udgjorde en stor udfordring i forbindelse med kalibrering af en strøm-...ings- og transportmodel, og der blev under projektet udviklet et partikel-...sportmodul til MIKE SHE modelsy-...met til beskrivelse af transporterede partiklers alder, "fødetid- og sted", transportveje osv. Dette modul viste sig specielt nyttigt i forbindelse med kalibreringen af strømnings- og transportdelen i forbindelse med SMP projekterne, men har endvidere siden vist sig utrolig velegnet til at kaste lys over nogle af de begreber, der anvendes i debatten omkring beskyttelseszoner.

## Begrebsdefinitioner

Indvindingsoplandet til en given indvindingsboring eller gruppe af indvindingsboringer (kildeplads) kan defineres, som det område i den filtersatte formation indenfor hvilket en tilstedeværende vandpartikel før eller siden vil nå hen til indvindingsboringen, se Figur 1. Dvs. dette område bestemmes alene på grundlag af strømningsforholdene i det magasin, hvor boringen er filtersat. Dette område fokuseres der fra forskellig side utroligt meget på, men området er jo egentligt kun interessant i forbindelse med en forureningssituation, hvor forureningen er nået helt ned i det aktuelle grundvandsmagasin.

Infiltrationsområdet for en given boring eller gruppe af boringer kan defineres som det område, hvor det oppumpede grundvand dannes via infiltration gennem jordens umættede zone eller via infiltration fra vådområder og vandløb, se Figur 1. Dette område kan sagtens - som det vil fremgå af det følgende – ligge langt fra boringerne og helt uden for indvindingsoplandet. Det er dette område, der er interessant i forbindelse med beskyttelsen af grundvandet, idet drikkevandskvaliteten direkte kan påvirkes af kvaliteten af det infiltrerende vand.

Det engelske begreb capture zone er i virkeligheden en kombination af indvindingsopland og infiltrationsområde, idet capture zone kan defineres som det (tre-dimensionale) område indenfor hvilket vandet strømmer til en given boring eller kildeplads. Her kan der altså være tale om et område, der strækker sig over flere geologiske lag, og specielt i forbindelse med forureningsundersøgelser er dette område interessant, se Figur 1.

### "Gummistøvleprojektet"

Miljøstyrelsen startede i 1993 en projektpakke, hvis formål blandt andet var at udvikle retningslinjer for den overordnede forvaltning af grundvandsressourcerne - det såkaldte gummistøvleprojekt. Et af projekterne i dette arbejde omhandlede metoder til udpegning af indvindingsoplande /1/. Formålet var at indsamle og vurdere forskellige metoder til bestemmelse af oplandet til en given indvinding, og derved give retningslinjer for en hurtig og realistisk udpegning af oplandene i forbindelse med vandressourceforvaltning. I denne publikation er flere forskellige, simple metoder til bestemmelse af indvindingsoplande beskrevet og metoderne er afprøvet på forskellige typiske danske lokaliteter.

Der er endvidere givet et ret akademisk eksempel på, hvorledes afstanden fra en given indvindingsboring til dennes "punktinfiltration" kan beregnes - altså infiltrationsområdet, hvor grundvandet til den givne boring primært dannes. Fælles for metoderne er, at de er tilnærmede, kun gælder under stationære forhold, og at de forudsætter kendskab til gradient-, transmissivitets-, lækage- og nettonedbørsforholdene i de berørte områder - størrelser, der ofte kun er kendt med endog meget store usikkerheder.

Alligevel kan det formentlig konkluderes, at disse metoder med en vis ret kan anvendes i forbindelse med mindre enkeltindvindinger i "ukomplicerede" geologiske og hydrogeologiske omgivelser.

### Regionale indvindingsoplande

Fyns Amt og Odense Vandselskab anmodede i sommeren 1996 Dansk Hydraulisk Institut (DHI) om at udpege indvindingsoplandene til de største kildepladser i området i nærheden af Odense i forbindelse med Amtets regionplankortlægning. DHI har gennem tid-

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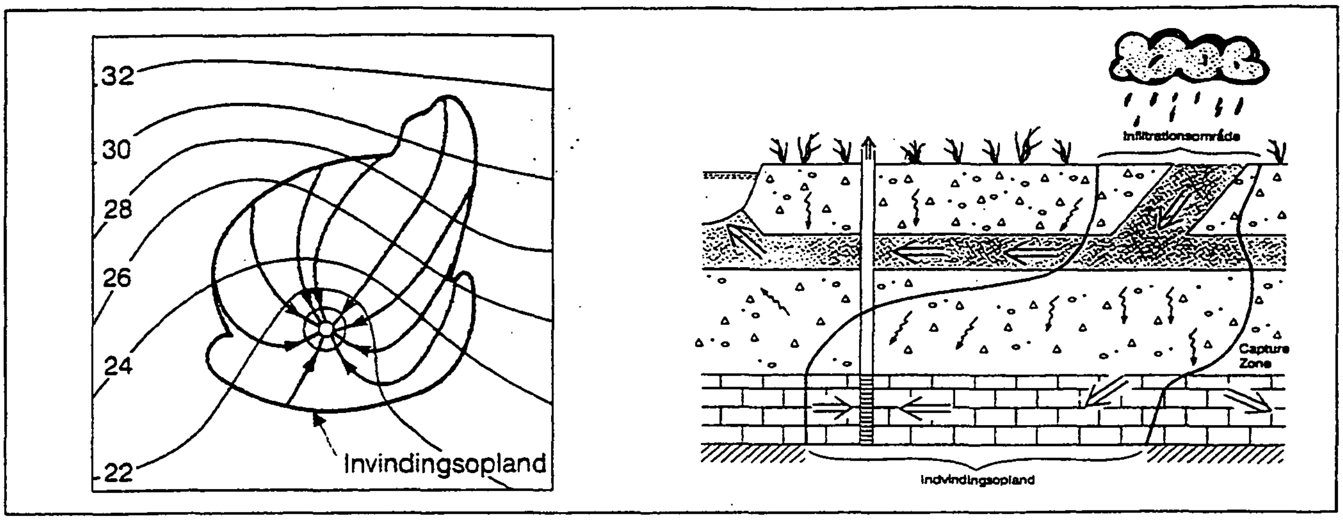



*Figur 1. Indvindingsopland, infiltrationsområde og "capture zone".*

ligere projekter opstillet, kalibreret og anvendt en MIKE SHE baseret hydrologisk model til vurdering af konsekvenser for grundvand og vandløb/vådområder af at etablere en ny kildeplads ved Nr. Søby /3/ og /4/. Det var derfor naturligt at tage udgangspunkt i denne model.

Modellen, som dækker det hydrologiske opland til Odense Å samt visse nabooplande - i alt ca. 1000 $km^2$ opløst i et 500 m beregningsnet - indeholder bl.a. en tre-dimensional beskrivelse af grundvandsstrømningerne. I den matematiske model er geologien simplificeret til en 5-lags model med et øvre og et nedre grundvandsmagasin adskilt af aquitarder og med et tyndt lag øverst, som indeholder den umættede zone. Modellen er kalibreret og valideret mod tidsserier af grundvandspotentialer og vandløbsafstrømninger for en 20-årig periode indeholdende både meget tørre og meget våde perioder. Den forventes som sådan at kunne give et meget realistisk billede af de faktisk forekommende strømningsforhold. Modellen indeholder ligeledes en egentlig beregning af "nettonedbøren" og de tids- og arealmæssigt varierende infiltrationsforhold.

Der er udpeget 15 kildepladser, hvortil indvindingsoplandene ønskes beregnet. Fra alle kildepladser foregår der en årlig indvinding på mere end 200.000 $m^3$. Beregningen af oplandene foregår med det nyudviklede partikeltransportmodul til MIKE SHE idet der først udvælges en repræsentativ periode, hvorfra strømningsberegningerne skal danne basis for de efterfølgende transportberegninger. I dette tilfælde er året 1994 valgt som et repræsentativt år, og transportberegningerne anvender gentagne gange årstidsvariationen i strømningsforhold. Herefter tildeles beregningselementerne et antal partikler, og under simuleringen bevæger disse sig med partikelhastigheden svarende til de aktuelle flux og porøsitetsforhold. Programmet holder styr på hvilke partikler, der havner i én af de udvalgte indvindingsboringer dvs. hvor kommer denne partikel fra og hvornår bliver den pumpet op. Resultatet af beregningen er således et tre-dimensionalt, tidsrelateret indvindingsopland (capture zone).

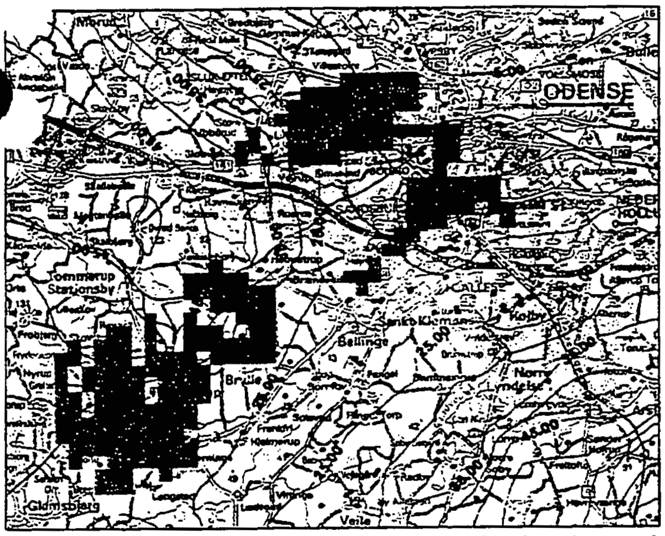
Konkret kan der ud fra beregningerne produceres figurer, der viser dels indvindingsoplandene til de enkelte kildepladser, se Figur 2, dels infiltrationsområderne, til de samme kildepladser, se Figur 3. Beregningerne er foretaget i en 75 års periode, hvilket svarer til, at der er opnået semi-stationære forhold med hensyn til capture zones.

Som det fremgår af Figur 2 er udstrækningen af indvindingsoplandene noget mere kompleks, end man umiddelbart ville forestille sig ved anvendelse af mere simple beregningsprincipper. I flere af oplandene er der "huller", hvilket skyldes, at lokale indvindinger og/eller lokale geologiske og hydrogeologiske forhold betinger, at vandet fra disse områder ikke strømmer til de udpegede indvindingsboringer. Oplandet til "Hovedværket" strækker sig uden om oplandet til "Eksercermarken" i en form, som de færreste ville kunne beregne eller optegne ud fra potentiale- og transmissivitetsforhold. I det hele taget vil det formentlig være umuligt at anvende simple metoder til bestemmelse af indvindingsoplande, når strømningsforholdene er styret af en kompliceret indbyrdes "kamp" om vandet mellem de forskellige kildepladser.

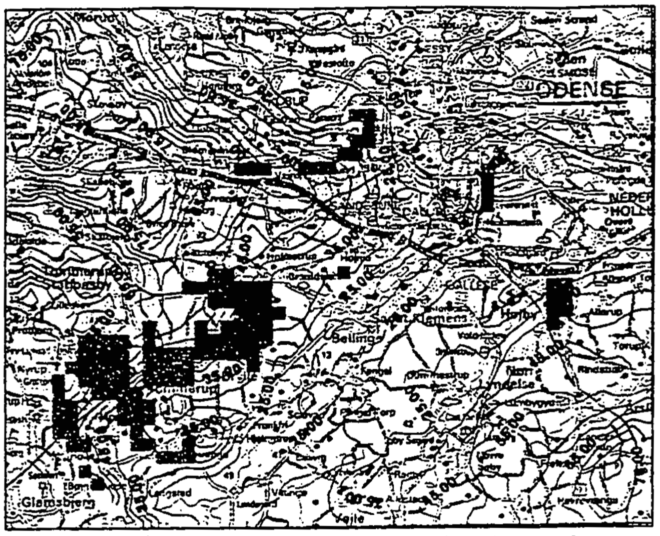
Figur 3 viser infiltrationsområderne, som her er defineret som de områder i det øverste beregningslag, hvor partiklerne finder ned til indvindingsboringerne. Det fremgår heraf, at disse områder er endnu mere komplekse, og har en langt mindre udstrækning, og i flere tilfælde ikke engang ligger inden for indvindingsoplandene. Det komplekse billede af infiltrationsområderne opstår bl.a., fordi der foregår strømninger i de øvre dele af grundvandszonen til vandløb og vådområder og til eventuelle lokale indvindingsboringer, som er filtersat i det øvre grundvandsmagasin.

## Lokale indvindingsoplande

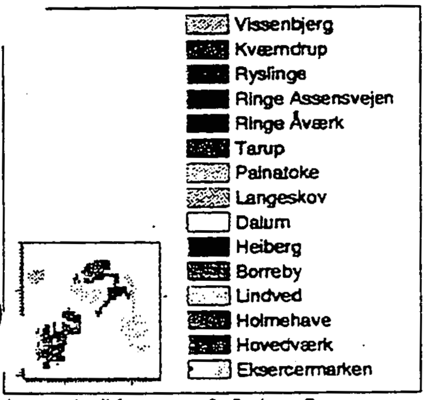
Birkerød Vandværk I/S anmodede i foråret 1996 ligeledes DHI om at vurdere en truende grundvandsforurening med henblik på at optimere afværgetiltagene og forudsige de fremtidige konsekvenser for Vandværket. Opgaven indebar bl.a. opstillingen af en grundvandsmodel for kildepladsen, som består af 10 boringer /5/. Modellen, som dækker det formodede indvindingsopland - i alt ca. 12 km2 opløst i et 50 m beregningsnet - er begrænset til at indeholde grundvandszonen og visse styrende overfladerecipienter. De kvartære aflejringer udgøres øverst af et 5 til 20 m tykt lag af moræneler, som underlejres af et regionalt sand-, grus- og siltlag med en tykkelse på op til 40 m. Disse aflejringer underlejres i visse dele af området af et tyndt lag af smeltevandsler, som har ringe vandgennemtrængelighed, mens det i andre områder direkte underlejres af prækvartære kalkaflejringer, som udgør det primære reservoir for vandindvinding.



Figur 2. Indvindingsoplande til de største kildepladser i nærheden af Odense.



Figur 3. Infiltrationsområder til kildepladserne i nærheden af Odense.




Legende til fugurene 2, 3, 4 og 5.

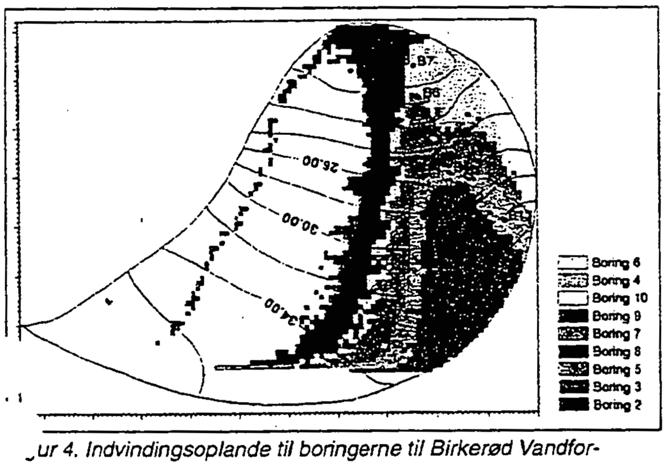
Modellen er i vertikalen opdelt i 10 beregningslag, hvilket giver en realistisk beregning af de tre-dimensionale strømningsforhold, der i høj grad er styret af indvindingsstrukturen, de omkringliggende recipienter (Sjælsø) og udbredelsen af det tynde lag af smeltevandsler. Der er anvendt samme fremgangsmåde som i Odense-projektet til bestemmelse af indvindingsoplande og infiltrationsområder.

Med den noget finere diskretisering af området i 50 m beregningselementer har det været muligt at bestemme indvindingsoplandene og infiltrationsområderne til de enkelte boringer på kildepladsen, som vist i Figur 4 og 5. Indvindingsoplandene viser et meget komplekst billede af, hvilke områder der bidrager med vand til de enkelte boringer - se fx indvindingsoplandene til B4, B5 og B9, som ud over at være "klemt" mellem andre oplande strækker uden om disse i nogle meget komplekse mønstre.

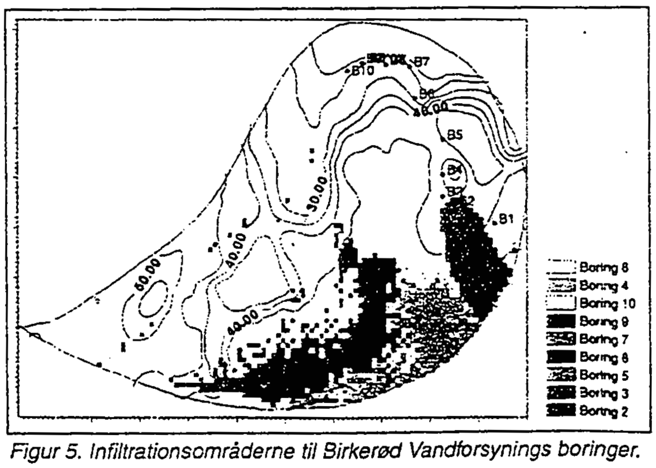
Infiltrationsområderne er ikke mindre komplekse, og er som det fremgår af Figur 5 beliggende langt fra selve indvindingsboringerne. Dette skyldes, at det vand, der infiltrerer på selve kildepladsen, strømmer via højtliggende sandlag direkte mod Sjælsø og dermed ikke bidrager til grundvandsdannelsen til de nedre magasiner på kildepladsen. Hvis man vil beskytte vandkvaliteten i indvindingsboringerne, skal man altså sikre, at det i infiltrationsområderne dannede grundvand har en god vandkvalitet. Det er altså i dette tilfælde lidt svært at se begrundelsen for at fokusere på den såkaldte kildepladszone, som er defineret som området mindre end 500 m fra indvindingsboringerne.

**Forureningssynderen**

Én af Birkerød Vandværks boringer er forurenet og modellen er anvendt til at sandsynliggøre beliggenheden af den eller de forurenende virksomhed(er). Figur 6 viser transporttider til den forurenede boring af et konservativt stof, som er infiltreret gennem den umættede grundvandszone. Sammenholdt med beliggenheden og driftsperioderne af mulige forureningskilder er "synderne" udpeget. Efterfølgende prøveboringer har med stor sandsynlighed bekræftet de opstillede hypoteser, og afværgetiltag tæt på forureningskilden er under projektering.




Figur 4. Indvindingsoplande til boringerne til Birkerød Vandforsyning.




Figur 5. Infiltrationsområderne til Birkerød Vandforsynings boringer.

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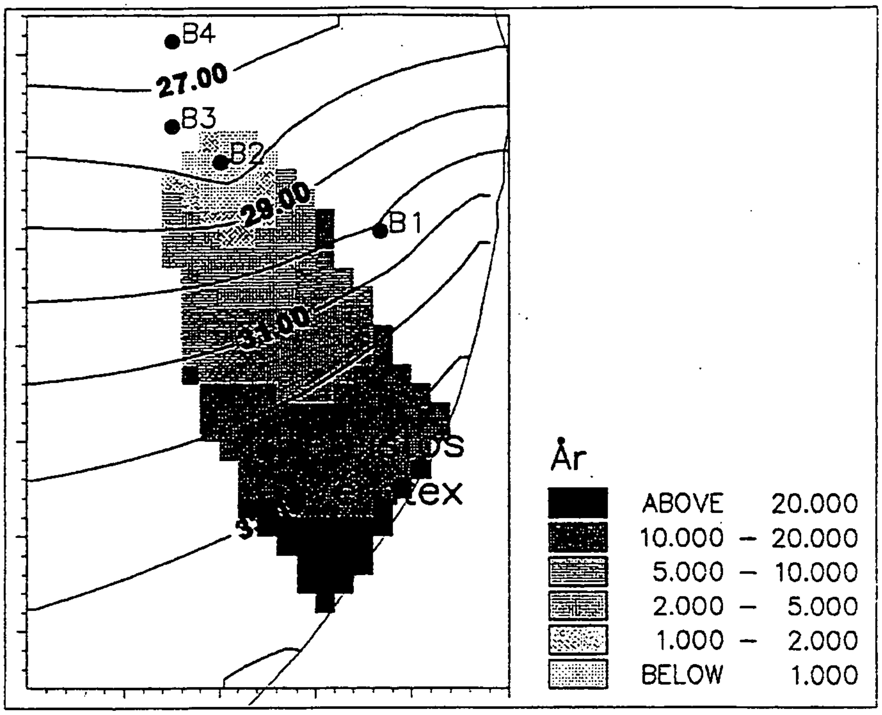


*Figur 6. Transporttider til en forurenet indvindingsboring.*

## Konklusioner

Ovenstående eksempler viser anvendeligheden af en numerisk partikeltransportmodel kombineret med en strømningsmodel i forbindelse med udpegningen af indvindingsoplande og infiltrationsområder. Det er formentlig det eneste redskab, der kan bestemme de komplekse områder under hensyntagen til aktuelle komplicerede forhold omkring geologi, hydrogeologi, indvindingsmønster osv. Selv om resultaterne - ligesom andre metoder - skal tages med de forbehold, som afspejles af usikkerhederne på bestemmelsen af de strømnings- og transportmæssige parametre i en numerisk model, kan dette være et vigtigt redskab i forbindelse med prioriteringen af indsatsen for en bedre grundvandskvalitet i fremtiden.

Partikelmodellen kan endvidere anvendes til at bestemme tidsrelaterede oplande til enkeltboringer, hvilket er en stor hjælp i forbindelse med udpegningen af forureningssyndere, idet disse informationer sammenholdt med informationer omkring mulige forureningskilder og perioder, hvor disse har kunnet forurene grundvandet, kan udpege forureningssyndere med stor sandsynlighed.

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ANDERS REFSGAARD, er ansat på Dansk Hydraulisk Institut (DHI) som afdelingsleder i Vandressourcedivisionen, hvor også JAN GREGERSEN, MICHAEL BUTTS og MICHAEL KRISTENSEN arbejder med anvendelsen af MIKE SHE modelsystemet i forbindelse med vandressource- og grundvandsforureningsundersøgelser.



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# Model and Data Requirements for Simulation of Runoff and Land Surface Processes

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## 1. Introduction

Hydrological models are usually applied under stationary climatological conditions and for purposes other than predicting the effects of climate change. Hence, important questions with regard to application of hydrological models within the global environmental change context are: (a) whether such models are applicable for this purpose; (b) which special test schemes are necessary in order to validate hydrological models for this purpose; (c) which special requirements this type of application puts to hydrological models; and (d) what are the data requirements for application of hydrological models in this context.

Traditionally, hydrological simulation models are classified in two main groups, namely the lumped, conceptual type and the distributed, physically based type. In Section 2, a brief description of three different hydrological models are given with respect to model structure and data requirements. Two of the models are typical representatives of the above two main groups, while the third one represents a mixed approach.

In Section 3, a test procedure recommended by Klemes (1985) is described briefly. This procedure is considered suitable for testing the applicability of hydrological models for predicting the effects of climate change.

In order to test the capability of different types of hydrological models to predict the hydrological regime under changed climatological conditions, a comprehensive and rigorous testing scheme has been carried out using three hydrological models on data from catchments in Zimbabwe. The data from Zimbabwe were chosen so that data sets

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both from very wet and very dry periods were available. The models were then subject to calibration in the wet period and subsequent validation in the dry period in the same catchment. The capability of the models to simulate runoff without prior calibration on data from the same catchment was also investigated. Some key results are presented in Section 4.

Section 5 presents views on how remote sensing data can most optimally be coupled with hydrological models and provide mutual benefits. Finally, in Section 6, a discussion is given on the requirements to hydrological models for coupling to Global Circulation Models.

## 2. Brief Description of Three Different Types of Hydrological Models

The following three models have been used for the Zimbabwe study:

- NAM: a lumped, conceptual rainfall-runoff model
- WATBAL: a semi-distributed hydrological model
- MIKE SHE: a fully distributed, physically based model

The NAM and MIKE SHE models can be characterized as very typical of their respective classes, while the WATBAL falls in between these two classes. All three models are being used on a routine basis at the Danish Hydraulic Institute (DHI) in connection with consultancy and research projects.

### *2.1 NAM*

NAM is a traditional hydrological model of the deterministic, lumped, conceptual type operating by continuously accounting for the moisture contents in four mutually interrelated storages. The NAM model was originally developed at the Technical University of Denmark (Nielsen and Hansen, 1973) and has been modified and applied extensively by DHI in a large number of engineering projects covering all climatic regimes of the world. Furthermore, the NAM has been transferred to more than 100 other organizations worldwide as part of DHI's MIKE 11 generalized river modeling package.

The structure of the model is illustrated in Figure 1. The input data consist of:

- Meteorological time series of precipitation on a daily or finer basis, potential evaporation on at least a monthly basis, and, if the snow routine is activated, temperature.
- A set of model parameters defining the hydrological characteristics of the catchment.
- Groundwater abstractions (optional).

In its present version, the NAM has a total of 17 parameters; however, in most cases, only about 10 of these are adjusted during calibration.

### 2.2 *WATBAL*

WATBAL was developed in the early 1980s by DHI in an attempt to provide a physically based, distributed hydrological model capable of using readily available data on land surface properties (topography, vegetation, soil), yet simple enough to allow large-scale applications within reasonable computational requirements. In the following, the model is introduced briefly, while more detailed information may be obtained in Knudsen et al. (1986).

WATBAL has been designed to account for the spatial and temporal variations of soil moisture. Based on distributed information on meteorological conditions, topography, vegetation, and soil types, the catchment area is divided into a number of hydrological response units as illustrated in Figure 2, with each unit being characterized by a different composition of the above features.

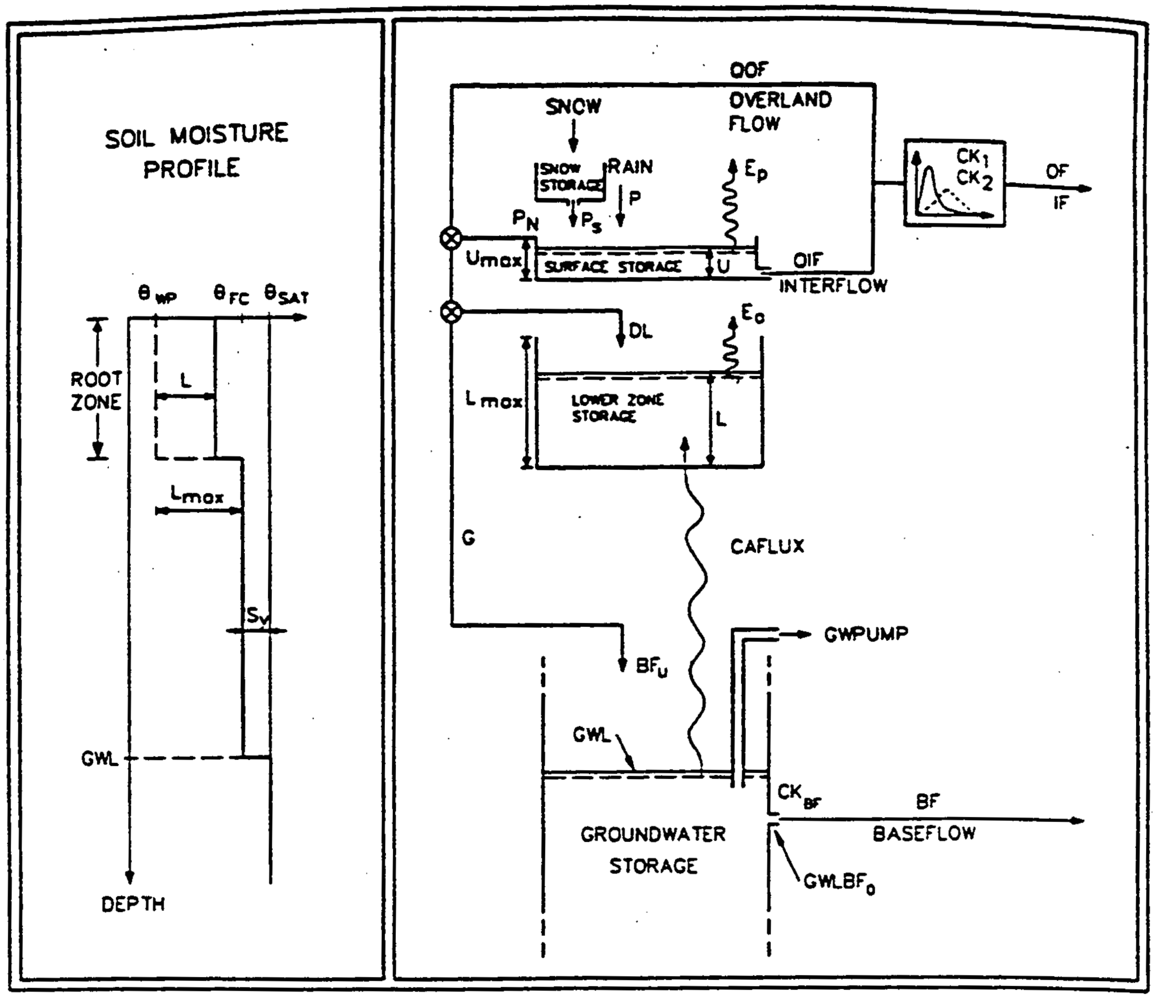


**Figure 1. Structure of the NAM model.**

These units are used to provide the spatial representation of soil moisture, while temporal variations within each unit are accounted for by means of empirical relations for the processes affecting soil moisture, using physical parameters particular to each unit.

For a description of subsurface flows, a simple lumped, conceptual approach is applied, using a cascade of linear reservoirs to account for the interflow and baseflow components (Figure 3).

In summary, WATBAL provides a physically based, distributed description of the processes affecting soil moisture (interception, infiltration, evapotranspiration, and percolation), while a conceptual approach is used to represent subsurface flows. In this way, WATBAL may be viewed as a compromise between limitations on data availability, the complexity of hydrological response at catchment scale, and the advantages of model simplicity.

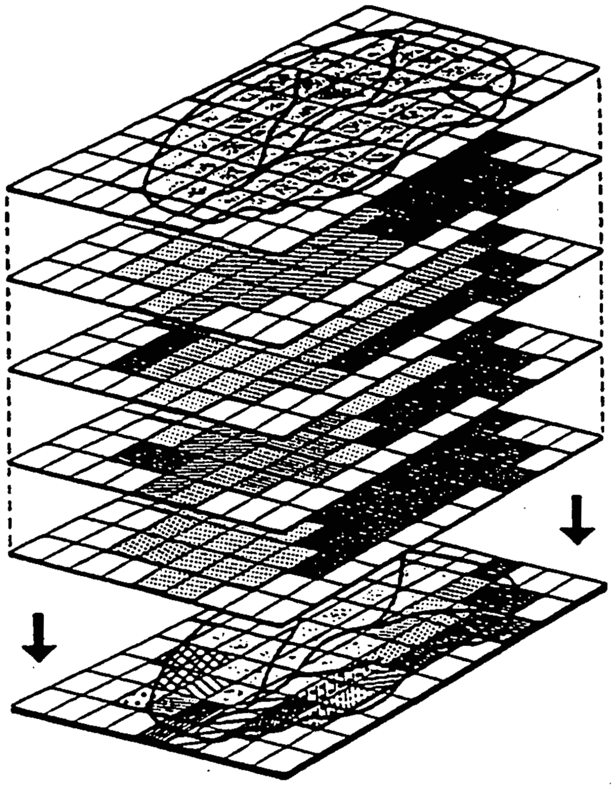


**Figure 2. WATBAL representation of catchment characteristics and definition of hydrological response units.**

The exogenous data required to operate the model comprise time series of rainfall on at least a daily basis and monthly evaporation data, while necessary model parameters are given in Table 1.

### *2.3 MIKE SHE*

The European Hydrological System (SHE) was developed in a joint effort by the Institute of Hydrology (UK), SOGREAH (France), and the Danish Hydraulic Institute (DHI). It is a deterministic, fully distributed, and physically based modeling system for describing the major flow processes of the entire land phase of the hydrological cycle. A description of the SHE is given in Abbott et al. (1986a,b). Since 1987, the SHE has been further developed independently by the three respective organizations which now are University of Newcastle (UK), Laboratoire d'Hydraulique de France, and DHI. DHI's version of the SHE, known as the MIKE SHE, represents significant new developments with respect to user interface, computational efficiency, and process descriptions.

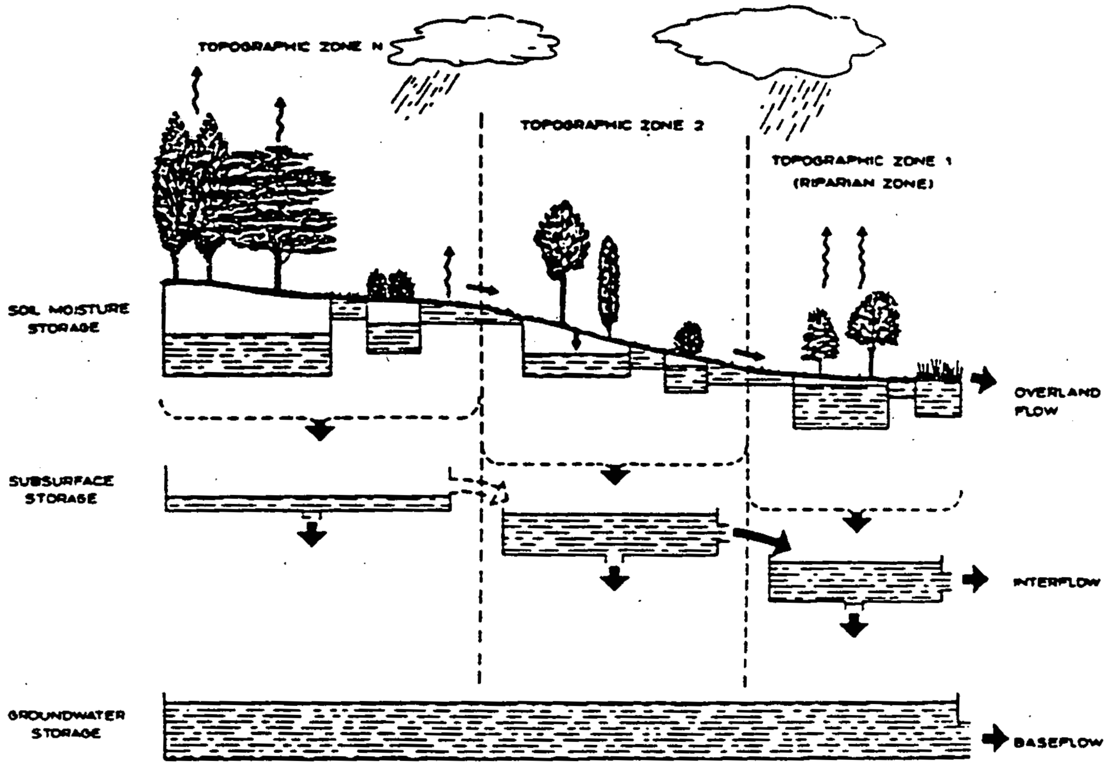


Figure 3. Principal structure of WATBAL.

MIKE SHE solves the partial differential equations for the processes of overland and channel flow and unsaturated and saturated subsurface flow. The model is completed by a description of the processes of snow melt, interception, and evapotranspiration. The flow equations are solved numerically using finite difference methods.

In the horizontal plane, the catchment is discretized in a network of grid squares. The river system is assumed to run along the boundaries of these. Within each square, the soil profile is described in a number of nodes which, above the groundwater table, may become partly saturated. Lateral subsurface flow is only considered in the saturated part of the profile. Figure 4 illustrates the structure of the MIKE SHE. A description of the methodology and some experiences of model application are presented in Refsgaard et al. (1992) and Jain et al. (1992).

Table 1. Model Parameters Required by WATBAL

| | |
|---|---|
| TOPOGRAPHY | Within each topographic zone: (depends on selected mode)<br>o length of flow plane<br>o slope<br>o manning number<br>o depression storage |
| VEGETATION | For each type of vegetation:<br>o leaf area index (time varying)<br>o root depth (time varying) |
| SOIL TYPES | For each texture class:<br>o wilting point<br>o field capacity<br>o total porosity<br>o saturated conductivity<br>o average suction |
| SUB-SURFACE REGIME | For each topographic zone:<br>o threshold value for interflow generation<br>o two time constants (interflow/percolation outlets)<br><br>Groundwater storage:<br>o groundwater area relative to catchment area<br>o time constant of base flow outlet |

The spatial and temporal variations in the catchment characteristics and meteorological input are provided in a series of two-dimensional matrices of grid square codes. A number of attributes describing either parametric data or input data is attached to each code. A list of the most important data and parameters is shown in Table 2.

The distributed description in the MIKE SHE allows the user to include and test against spatially varying data. MIKE SHE is a multioutput model which, besides discharge in any river link, also produces information about water table elevations, soil moisture contents, infiltration rates, evapotranspiration, etc. in each grid square.

MIKE SHE is usually categorized as a physically based model. Strictly speaking, the characterization is only correct if the model is applied on an appropriate scale. A number of scale problems arise when the MIKE SHE is set up on a regional scale. In addition, if there is a considerable uncertainty attached to the basic information, and if the spatially and temporal varying variables (such as groundwater table elevations) cannot be validated against observations, MIKE SHE is, on that particular setup, not physically based but rather a detailed conceptual model. In this case, the calibration procedure is usually to adjust the parameters with the largest uncertainties attached, within a reasonable range.

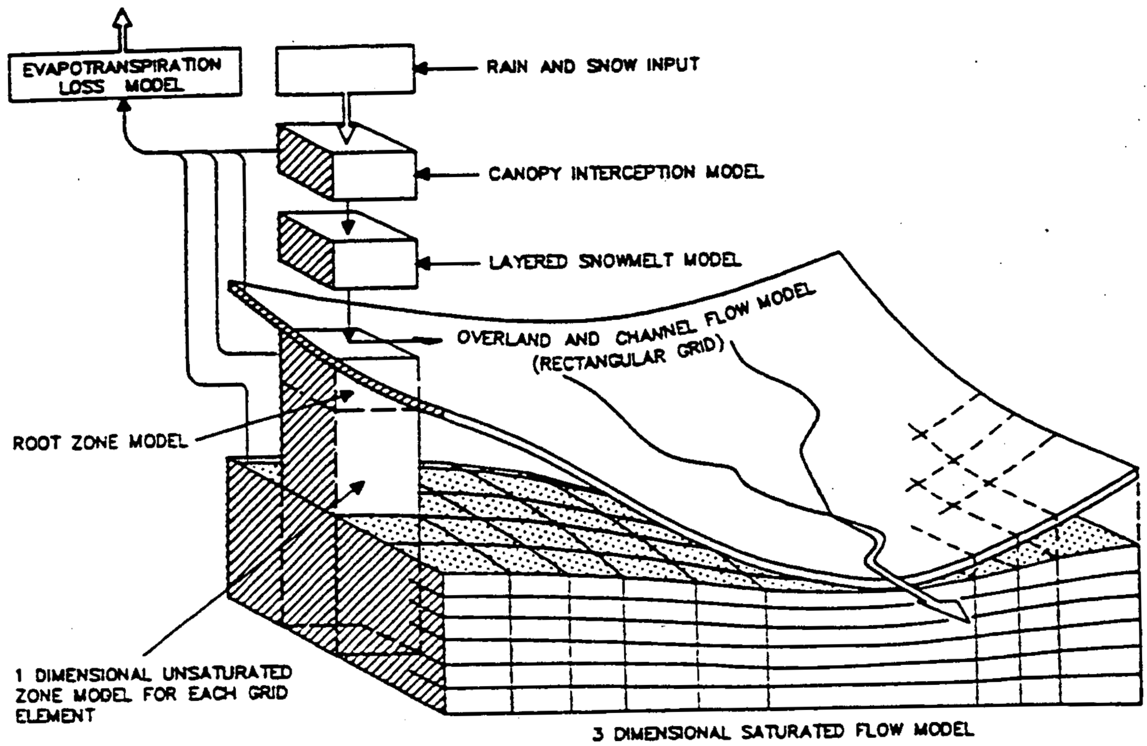


Figure 4. Schematic presentation of the MIKE SHE.

An illustration of the outputs simulated by a distributed model like the MIKE SHE is given in Figure 5. In this figure, selected results from a two-year simulation of the 422 km$^2$ Karup catchment are shown. To the left, the spatial distribution of actual evapotranspiration, vegetation type, and depth to groundwater table on July 6, 1976 is shown. The right column shows the temporal variation of rainfall input, potential and actual evapotranspiration, leaf area index, and soil moisture content in the upper 1.5 m of one of the 500 m x 500 m grids.

**Table 2. Important Data and Parameters Required by the MIKE SHE**

| | |
|---|---|
| GENERAL | For each grid square or river link:<br>o topography<br>o impermeable bed<br>o channel geometry<br>o meteorological station code<br>o vegetation type code<br>o soil profile code |
| VEGETATION | For each vegetation type:<br>o leaf area index (time varying)<br>o rooting depth (time varying) |
| SOIL (unsaturated zone) | For each soil type:<br>o retention curve<br>o hydraulic conductivity function |
| SOIL (saturated zone) | Saturated hydraulic conductivity in horizontal and vertical directions:<br>o storage coefficients<br>o drainage depth<br>o drainage time constant |
| OTHER | o roughness coefficients in each grid square (surface) and for river links<br>o groundwater abstraction data |

## 3. Testing Schemes for Validation of Hydrological Models

For a long time, the testing of hydrological models through validation on independent data has been emphasized by the World Meteorological Organization (WMO). In their pioneering studies (WMO, 1975, 1986), several hydrological simulation models were tested on the same data from different catchments. The actual testing, however, only included the standard split-sample test comprising an initial calibration of the model and subsequent validation based on data from an independent period.

WMO and UNESCO have included a project for "Testing the transferability of hydrological simulation models" (Project D.5) as part of the World Climate Programme--Water, WMO (1985). The study by DHI (1993) may be viewed as a contribution to the WCP-Water/D.5-project.

The hierarchial testing scheme of Klemes (1985) appears to be suitable for testing the capability of a model to predict the hydrological effects of climate change. Klemes (1985) distinguished between simulations conducted at the same station (catchment) used for calibration and simulations conducted for ungauged catchments. He also distinguished between cases where climate and land use are stationary and cases where they are not.

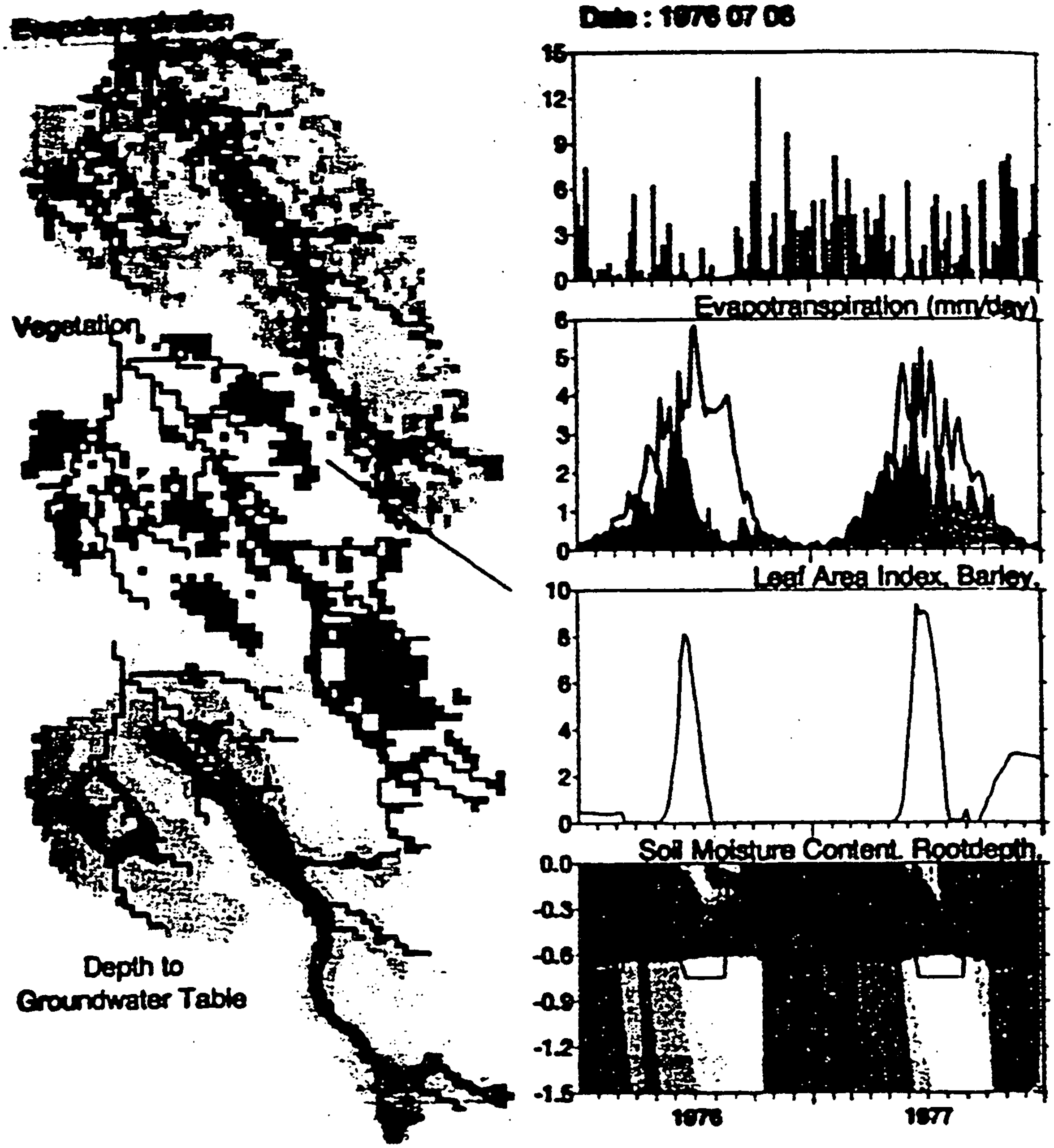


Figure 5. Illustration of selected output generated by MIKE SHE for the 422 km² Karup catchment in Denmark.

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This combines to the definition of four basic categories of typical modeling tests:

(a) Split-sample test: Calibration of model based on 3-5 years of data and validation on another period of similar length.

(b) Differential split-sample test: Calibration of model based on data before catchment change occurred, adjustment of model parameters to characterize the change, and validation on the subsequent period.

(c) Proxy-basin test: No direct calibration allowed, but advantage of information from other gauged catchments may be taken. Hence, validation will comprise identification of a gauged catchment deemed to be of similar nature as the validation catchment, initial calibration, transfer of model including adjustment of parameters to reflect actual conditions within validation catchment, and validation.

(d) Proxy-basin differential split-sample test: Again, no direct calibration is allowed, but information from other catchments may be used. Thus, validation will comprise initial calibration on other relevant catchment, transfer of model to validation catchment, selection of two parameter sets to represent the period before and after the change, and subsequent validations on both periods.

**4. Test of Hydrological Models for Runoff Simulations Under Varying Climate Conditions in Zimbabwe**

The following is based on results from a research project conducted at Danish Hydraulic Institute (DHI, 1993).

### *4.1 Selected Catchment*

The three catchments which have been selected for the model tests in Zimbabwe are Ngezi-South, Lundi, and Ngezi-North. The locations of the catchments are shown in Figure 6.

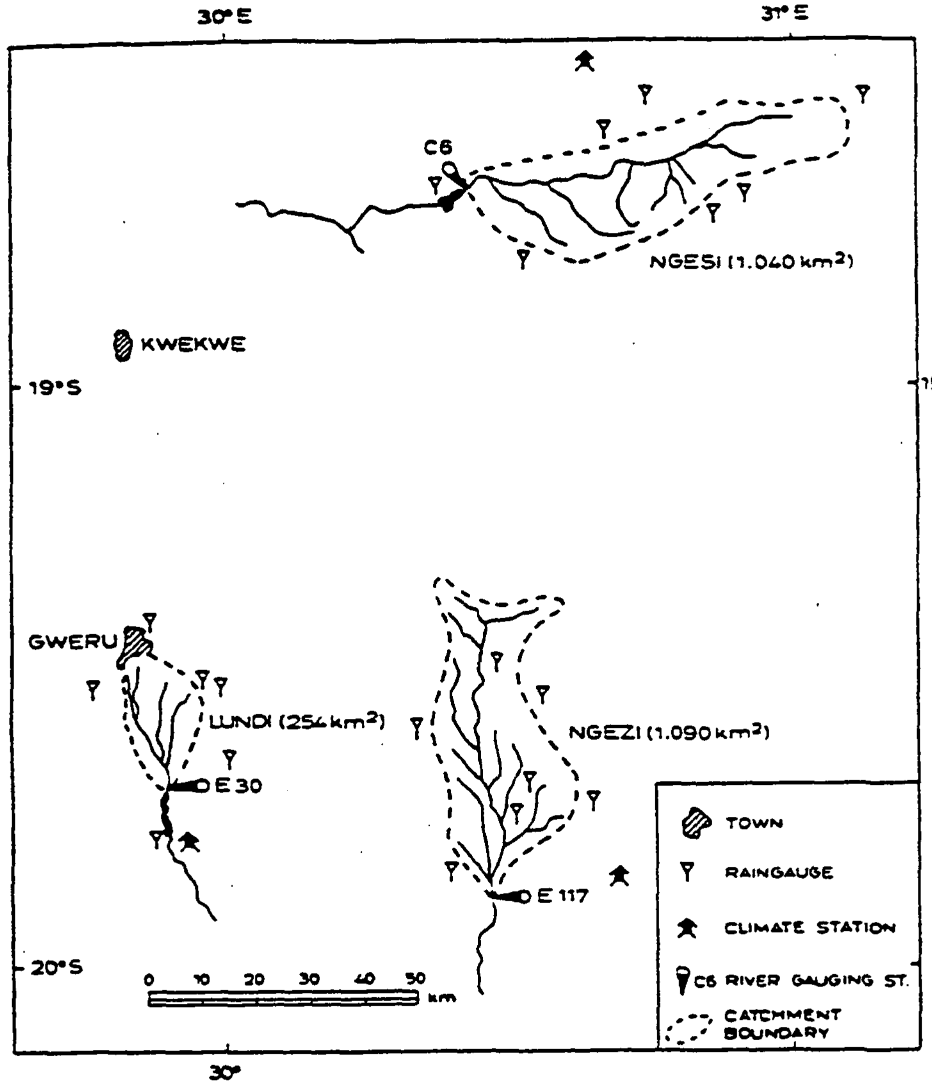


**Figure 6. Locations of the three catchments in Zimbabwe.**

A brief data collection/field reconnaissance to Zimbabwe was arranged to obtain relevant information. This included basic hydrometeorological data comprising daily series of rainfall and runoff and monthly series of pan evaporation. Detailed information on land use was obtained through subcontracting the University of Zimbabwe to prepare land use maps based upon 1:25,000 aerial photographs, while information on soil and vegetation characteristics was collected from various institutions and relevant literature. Furthermore, 1:50,000 topographical maps were collected and digitized.

Finally, available data on vegetation characteristics, soil characteristics, hydrogeology, and water rights were obtained. A more detailed description is given in DHI (1993).

Some of the key features for the three catchments are presented in Tables 3 and 4. It is noticed from the rainfall and runoff figures in Table 3 that there are very large variations between some of the periods. From Table 4, it appears that there are significant differences in the vegetation and soil characteristics from catchment to catchment.

**Table 3. Catchment Areas, Modeling Periods, Rainfall, and Runoff Values for the Three Zimbabwean Test Catchments**

| Catchment | Area (km2) | Periods | Average rainfall (mm/year) | Average runoff (mm/year) |
|---|---|---|---|---|
| Ngezi-South | 1090 | 1971/72 - 74/75 | 896 | 179 |
| | | 1975/76 - 78/79 | 873 | 128 |
| Lundi | 254 | 1971/72 - 75/76 | 915 | 171 |
| | | 1981/82 - 83/84 | 497 | 8 |
| Ngezi-North | 1040 | 1977/78 - 78/79 + 81/82 - 83/84 | 607 | 47 |

### *4.2 Testing Scheme*

The testing scheme for the Zimbabwe study carried out in DHI (1993) is illustrated in Table 5. The testing of the involved models has been undertaken in parallel and in the following sequence:

A: Split-sample test based on data from Ngezi-South comprising an initial calibration of the models and a subsequent validation using data for an independent period.



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C: Proxy-basin test. Transfer of models to the Lundi catchment, adjustment of parameters to reflect the prevailing catchment characteristics, and validation without any calibration.

(C): Same as above, but adjusted by allowing model calibration based on one year of runoff data.

B: Differential split-sample test. Model calibration based on data from an initial calibration period and validation based on data from a subsequent period. The differential nature of this test is justified by the fact that the later independent period includes three successive years (1981/82-1983/84) with a markedly lower rainfall than otherwise and, hence, represents a nonstationary climate scenario.

D: Proxy-basin differential split-sample test. Transfer of models to the Ngezi-North catchment, adjustment of parameters to represent the catchment characteristics, and validation by runoff simulation over a nonstationary climate period.

(D): Same as above, while allowing models to be calibrated using a short-term (one year) record.

In this paper, only the results from tests A, C, and B are presented.

### *4.3 Performance Criteria*

For measuring the performance of the models for each test, a standard set of criteria has been defined. The criteria have been designed with the sole purpose of measuring how closely the simulated series of daily flows agree with the measured series. Due to the generalized nature of the defined model validations, it has been necessary to introduce several criteria for measuring the performance with regard to water balance, low flows, and peak flows.

The standard set of performance criteria comprises a combination of graphical plots and numerical measures. The graphical diagrams used include joint plots of the simulated and observed hydrographs, scatter diagram of monthly runoffs, flow duration curves, and scatter diagram of annual maximum discharges. To support the graphical presentations, various numerical measures are computed, including the overall water balance, the Nash-Sutcliffe coefficient (R2), and an index (EI) measuring the agreement between the simulated and observed flow duration curves. Furthermore, additional measures for each hydrological year are computed.

**Table 4. Land Use Vegetation and Soil Characteristics Estimated from Available Information and a Brief Field Visit**

| | Ngezi-South | Lundi | Ngezi-North |
|---|---|---|---|
| Land use/vegetation (area %) | | | |
| Dense/closed woody vegetation | 7 | 13 | 10 |
| Open woody vegetation | 36 | 25 | 35 |
| Sparse woody vegetation | 14 | 19 | 14 |
| Grassland | 11 | 39 | 16 |
| Cropland | 29 | 3 | 19 |
| Abandoned cropland | 2 | 0 | 6 |
| Rock outcrops | 1 | 0 | 0 |
| Soil depth range (m) | 0 - 2.5 m | 0 - 1 m | 0.5 - 6 m |
| Saturated hydraulic conductivity in root zone soil (mm/hour) | Range: 1 - 250<br>Average: 80 | Range: 1 - 70<br>Average: 60 | Range: 2-100<br>Average: 50 |
| Available water content in root zone soil (vol %) | Range: 10 - 14<br>Average: 12 | Range: 10 - 12<br>Average: 11 | Range: 9 - 29<br>Average: 17 |

The coefficient R2, introduced by Nash and Sutcliffe (1970), is computed on the basis of the sequence of observed and simulated monthly flows over the entire testing period (perfect agreement for R2 = 1):

$$R2 = 1 - \sum_{m=1}^{M} \left(Q_m^o - Q_m^s\right)^2 / \sum_{m=1}^{M} \left(Q_m^o - \bar{Q}^o\right)^2$$

where:



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$M$ = total number of months;

$Q_m^s$ = simulated monthly flows;

$Q_m^o$ = observed monthly flows; and

$\bar{Q}^o$ = average observed monthly flows over the entire period.

**Table 5. Model Validations, Zimbabwe**

| Catchment/G.S.No. | Valid. Type | 71/72 | | 73/74 | | 75/76 | | 77/78 | | 79/80 | | 81/82 | | 83/84 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ngezi South/E117 | A | ▒ | ▒ | ▒ | ▒ | █ | █ | █ | █ | | | | | |
| Lundi/E30 | C | █ | █ | █ | █ | █ | | | | | | | | |
| | (C) | █ | █ | █ | █ | ▒ | | | | | | | | |
| | B | ▒ | ▒ | ▒ | ▒ | ▒ | | | | | | █ | █ | █ |
| Ngezi North/C6 | D | | | | | | | █ | █ | █ | █ | █ | █ | █ |
| | (D) | | | | | | | ▒ | █ | █ | █ | █ | █ | █ |

A: Split-sample test
B: Differential split-sample test
C: Proxy-basin test
D: Proxy-basin differential split-sample test

▒ Model calibration, i.e. adjustment of model parameters to fit observed hydrograph
█ Model validation, i.e. no calibration allowed; comparison with observed hydrograph after simulation

The flow duration curve error index, EI, provides a numerical measure of the difference between the flow duration curves of simulated and observed daily flows (perfect agreement for EI = 1):

$$EI = 1 - \int \left[ f_o(q) - f_s(q) \right] dq \, / \int f_o(q) \, dq$$

where:

$f_o(q)$ = flow duration curve based on observed daily flows; and

$f_s(q)$ = flow duration curve based on simulated daily flows.

Thus, the above criteria measure the extent to which the models are able to provide an accurate representation of the overall water balance, whether major discrepancies for individual years occur, the overall accuracy of the simulated series of monthly flows, and its capability to represent the overall pattern of the daily flows, that is, the frequency of occurrence of low, medium, and high flows. In spite of its incompleteness, the above criteria provide a reasonable summary of the overall model performance.

### *4.4 Results of Model Validations*

(a) Split-sample test: This test is based on data from Ngezi-South and comprises an initial calibration of the models and a subsequent validation using data for an independent period. The main results are summarized in Table 6.

As indicated by this table, the performance of all three models is generally very similar. All models are able to provide a close fit to the recorded flows for the calibration period while, for the independent validation period, the performance is somewhat reduced as expected. The reduction is, however, limited, and all models are able to maintain a very good representation of the overall water balance, the interannual and seasonal variations, as well as the general flow pattern.

(c) Proxy-basin test (ungauged catchment): This test comprises a transfer of models to the Lundi catchment, adjustment of parameters to reflect the prevailing catchment characteristics, and validation without any calibration.

The proxy-basin test was arranged to test the capability of the different models to represent runoff from an ungauged catchment area; hence, no calibration was allowed prior to the simulation. For all models, three alternative runoff simulations were prepared, reflecting an expected low, central, and high estimate, respectively.

All models have used the experience from the Ngezi-South calibrations in combination with the available information on the particular catchment characteristics for Lundi. While the NAM model has used this information in a purely subjective manner to revise model parameters, both the WATBAL and MIKE SHE models have used this

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information directly for the model setup. The estimates prepared by the latter two models have, however, also been influenced by the individual modelers subjective interpretation of the available information on soil and vegetation characteristics.

Table 6. Ngezi-South. Summary of Split-Sample Test Results. All flows are in mm/year.

| CALIBRATION | | | QSIM - QOBS | | |
|---|---|---|---|---|---|
| Year | Rain | QOBS | NAM | WATBAL | MIKE SHE |
| 1971/72 | 890 | 131 | -19 | -11 | -11 |
| 72/73 | 317 | 2 | 1 | - 1 | 7 |
| 73/74 | 1,290 | 349 | 14 | - 5 | -10 |
| 74/75 | 1,087 | 236 | 5 | 8 | 2 |
| Mean: | 896 | 179 | 0 | - 2 | - 3 |
| Mean \| QSIM - QOBS\| : | | | 10 | 6 | 8 |
| R2 | | | .97 | .96 | .97 |
| EI | | | .88 | .95 | .91 |

| VALIDATION | | | QSIM - QOBS | | |
|---|---|---|---|---|---|
| Year | Rain | QOBS | NAM | WATBAL | MIKE SHE |
| 1975/76 | 879 | 90 | 3 | 24 | 5 |
| 76/77 | 872 | 116 | -28 | 2 | 7 |
| 77/78 | 1,131 | 245 | 26 | 37 | 67 |
| 78/79 | 609 | 59 | -13 | 2 | - 9 |
| Mean: | 873 | 128 | - 3 | 16 | 18 |
| Mean \| QSIM - QOBS\| : | | | 18 | 16 | 22 |
| R2 | | | .89 | .86 | .84 |
| EI | | | .74 | .86 | .80 |

As an example of the model performance, the hydrograph simulated by the MIKE SHE is shown in Figure 7 for two of the years, together with the flow duration curves and the scatter diagram of monthly discharges for the entire five year validation period. A summary of the main results of the proxy-basin tests is given in Table 7.

In general, all models provide an excellent representation of the general flow pattern yet with some discrepancies for the small and/or larger peaks. As seen in Table 7, the best

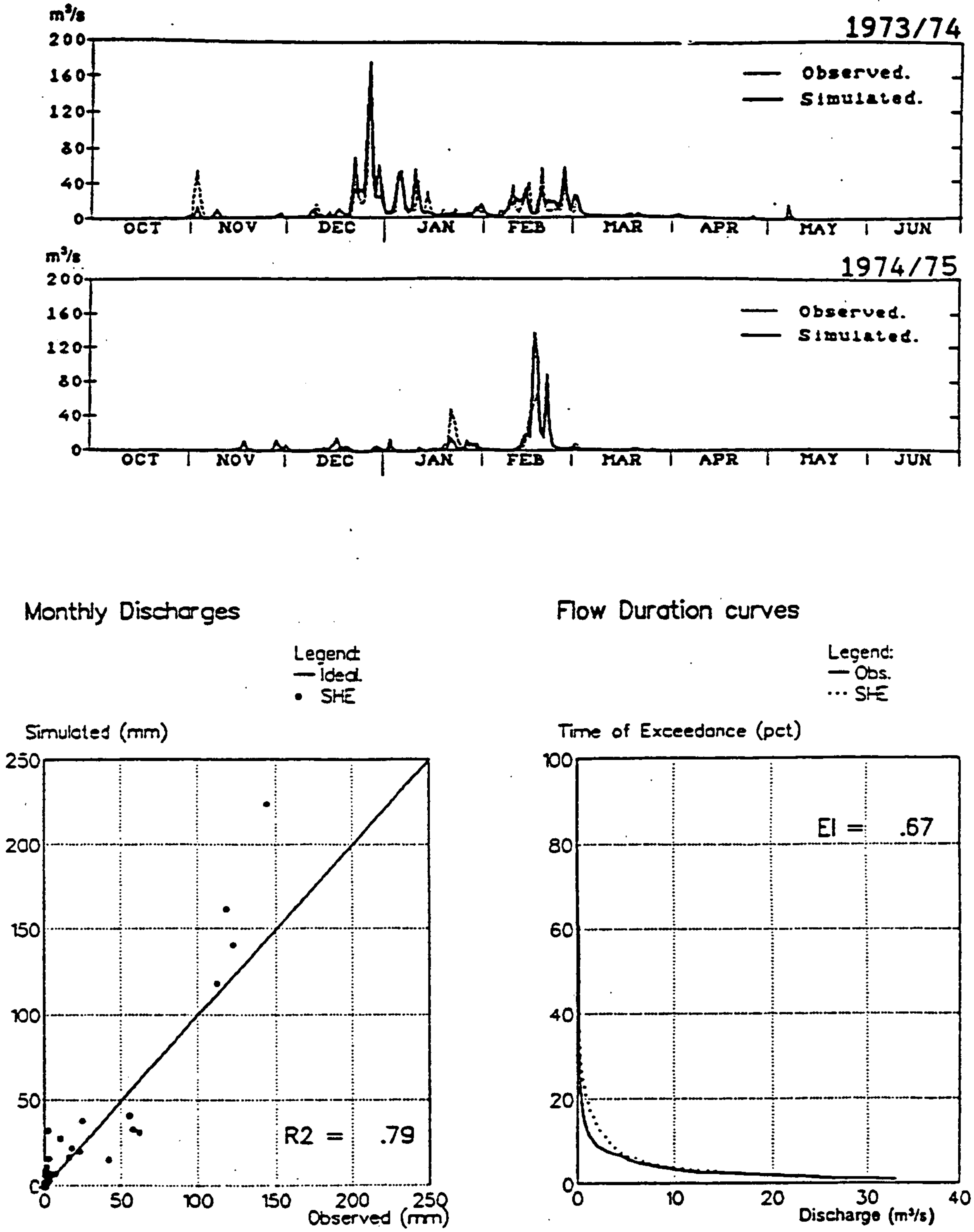


**Figure 7. Lundi MIKE SHE (central estimate) proxy-basin test hydrographs from two of the years, together with flow duration curves and scatter diagram of monthly discharges for the entire period.**

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runs of the individual models provide a good representation of the overall water balance, while maintaining the significant interannual variability to a satisfactory degree and an overall good simulation of flows within individual months.

Table 7. Lundi. Summary of Proxy-Basin Test Results. All flows are in mm/year.

| PROXY-BASIN TEST | | | QSIM - QOBS | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | NAM | | | WATBAL | | | MIKE SHE | | |
| Year | Rain | QOBS | Low | Ctr. | High | Low | Ctr. | High | Low | Ctr. | High |
| 1971/72 | 920 | 89 | 5 | 63 | 122 | 1 | 3 | 32 | - 7 | 27 | 29 |
| 72/73 | 371 | 2 | 2 | 5 | 10 | - 1 | - 1 | 6 | 10 | 21 | 21 |
| 73/74 | 1,384 | 460 | -44 | 72 | 151 | - 78 | 3 | 34 | 43 | 98 | 120 |
| 74/75 | 1,046 | 217 | -36 | 44 | 116 | -55 | -17 | 14 | 8 | 45 | 61 |
| 75/76 | 857 | 89 | -23 | 23 | 77 | -14 | -11 | 11 | -20 | 13 | 22 |
| Mean: | 915 | 171 | -19 | 42 | 95 | -29 | - 5 | 19 | 7 | 41 | 51 |
| Mean \| QSIM - QOBS \| : | | | 22 | 41 | 95 | 30 | 7 | 19 | 18 | 41 | 51 |
| R2 | | | .89 | .87 | .57 | .85 | .91 | .90 | .86 | .79 | .71 |
| EI | | | .66 | .72 | .41 | .59 | .76 | .75 | .78 | .67 | .63 |

The overall performance of the central estimates by the NAM and MIKE SHE models is somewhat reduced compared to validation runs for the Ngezi-South catchment as expected when no calibration is possible. The estimates would, however, still be very valuable for all practical purposes. For the WATBAL model, the central estimate is even better than obtained for the validation period for Ngezi-South, providing for a very accurate representation of observed runoff record.

(b) Differential split-sample test (nonstationary climate): This test consists of model calibrations based on data from four wet years (1971-76) and validation on data from three very dry years (1981-84). The purpose of this test is to assess the capability of the models to do simulations during nonstationary time periods.

As an example of the model performance, the hydrograph simulated by the MIKE SHE is shown in Figure 8 for two of the years, together with the flow duration curves and scatter diagram of monthly discharges for the entire five-year validation period. A summary of the main results of the differential split-sample tests is given in Table 8.

Table 8. Lundi. Summary of Differential Split-Sample Test Results. All flows are in mm/year.

| CALIBRATION | | | QSIM - QOBS | | |
|---|---|---|---|---|---|
| Year | Rain | QOBS | NAM | WATBAL | MIKE SHE |
| 1971/72 | 920 | 89 | 23 | 14 | -16 |
| 72/73 | 371 | 2 | - 1 | 1 | - 1 |
| 73/74 | 1,384 | 460 | - 3 | 14 | 28 |
| 74/75 | 1,046 | 217 | - 3 | - 5 | 3 |
| 75/76 | 857 | 89 | -17 | - 1 | -29 |
| Mean: | 915 | 171 | 0 | 5 | - 3 |
| Mean \| QSIM - QOBS\| : | | | 9 | 7 | 15 |
| R2 | | | .92 | .92 | .86 |
| EI | | | .82 | .80 | .82 |

| VALIDATION | | | QSIM - QOBS | | |
|---|---|---|---|---|---|
| Year | Rain | QOBS | NAM | WATBAL | MIKE SHE |
| 1981/82 | 416 | 10 | - 2 | 4 | - 6 |
| 82/83 | 528 | 7 | - 3 | 9 | 0 |
| 83/84 | 547 | 7 | 2 | 8 | 1 |
| Mean: | 497 | 8 | - 1 | 7 | - 2 |
| Mean \| QSIM - QOBS\| : | | | 2 | 7 | 2 |
| R2 | | | .66 | -1.65 | .13 |
| EI | | | .55 | - .02 | .56 |

As evident from the plots of the associated hydrographs, the MIKE SHE is capable of reproducing the intermittent pattern of the measured hydrograph during the very dry period comprising a small number of very small peaks and intermediate periods with nil

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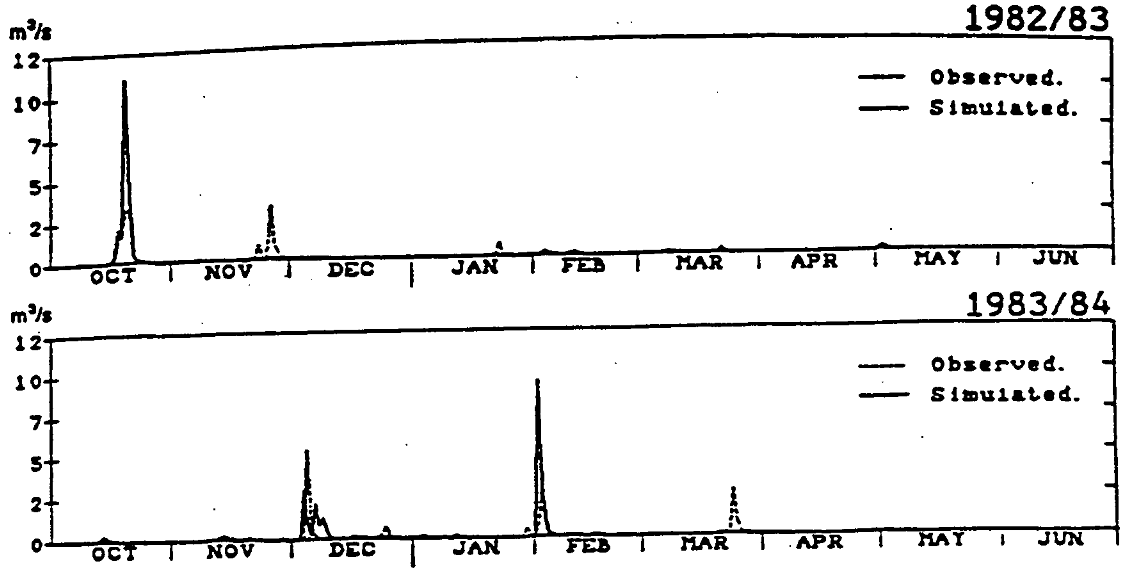


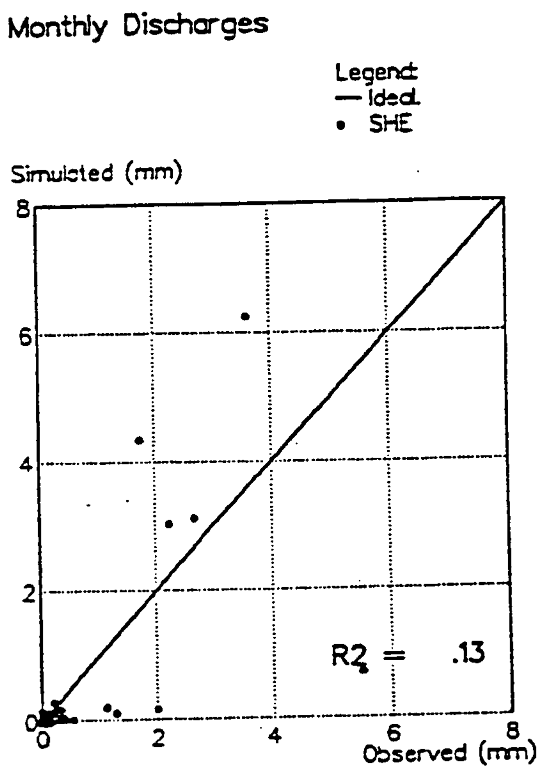


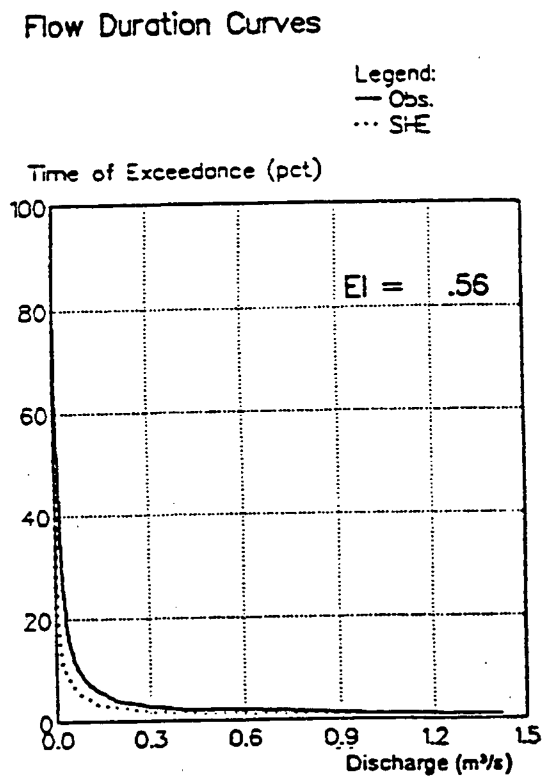


**Figure 8. Lundi MIKE SHE differential split-sample test hydrographs from two of the validation years together with flow duration curves and scatter diagram of monthly discharges for the entire validation period.**

or insignificant flows only. The same conclusion applies in general to the other two models (DHI, 1993).

The WATBAL model, however, grossly overestimates the peaks in the relative sense, causing the simulated average runoff to be about twice of that measured (15 mm compared to 8). For the NAM and MIKE SHE models, a much better performance is obtained, providing a quite accurate representation of the measured record in general. The related statistics are poorer than in the other testing schemes, but it should be noted that even small deviations causes poor statistics when mean flows are as low as in this case.

### *4.5 Conclusions on the Model Validations on the Zimbabwe Catchments*

The following conclusions are based primarily on the results shown above; however, some of the other results of the original research project (DHI, 1993) are also referred to.

In view of the difficult tasks given to the models involving simulation for ungauged catchments and nonstationary time periods, the overall performance of the models is considered quite impressive. The overall water balance agrees within $\pm 25\%$ in all cases but one, and good results are achieved without balancing out excessive positive and negative deviations within individual years. In most cases, the models score an R2 value at about 0.8 or greater and an EI-index generally above 0.7.

In addition to the above general comments, the following is noted with regard to the specific types of validations undertaken.

For the split-sample test, the NAM, WATBAL, and MIKE SHE models generally exhibit similar performance. All models are able to provide a close fit to the recorded flows for the calibration period, without severely reducing the performance during the independent validation period. Hence, this test suggests that, if an adequate runoff

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period for a few (3-5) years exists, any of the models could be used as a reliable tool for filling in gaps in such records or used to extend runoff series based on long-term rainfall series. Considering the data requirements and efforts involved in the setup of the different models, however, a simple model of the NAM type should generally be selected for such tasks.

For the proxy-basin tests, designed for validating the capability of the models to represent flow series of ungauged catchments, it had been expected that the physically based models would produce better results than the simple type of models. The results, however, do not provide unambiguous support for this hypothesis.

All three models generated good results, with the WATBAL providing slightly more accurate results than the others. Therefore, for the Zimbabwean conditions, the additional capabilities of the MIKE SHE as compared to the WATBAL, namely the distributed, physically based features relating to subsurface flow, proved to be of little value in simulating the water balance.

For the proxy-basin tests, it is noticed that the uncertainty range represented by the low-high estimates is significantly larger for the NAM than for the WATBAL and MIKE SHE cases. This probably reflects the fact that parameter estimation for ungauged catchments is generally more uncertain for the NAM, the parameters of which are semi-empirical coefficients without direct links to catchment characteristics.

A general experience of the modified proxy-basin tests is that allowing for model calibration based on only one year of runoff data improves the overall performance of all models. The improvement appears to be particularly significant for the NAM model, which also showed the largest uncertainties in the cases where no calibration was possible.

For the differential split-sample tests, all models have been able to simulate flows of the right order of magnitude and correct pattern. Thus, all models have proved to be able to simulate the runoff pattern in periods with much reduced rainfall and runoff as compared to the calibration period.

On the basis of these results, there appears no immediate justification for using an advanced type of model to represent flows following a significant change of rainfall, providing a number of years as available for calibration purposes. It is tempting to extend this finding to suggest that the simple type of model could be used to assess the impact of climate change on water resources. However, it should be recognized, that the above results cannot fully justify such a hypothesis, because a long-term climate change would probably bring about changes in vegetation and their evaporation. This type of unstationarity has not been adequately tested.

In summary, the results of the comprehensive validations suggest that, given a few years of runoff measurements, a lumped model of the NAM type would be a suitable tool from the point of view of technical and economical feasibility. This applies for catchments with homogeneous climatic input as well as cases where significant variations in the exogenous input are encountered.

For ungauged catchments, however, where accurate simulations are critical for water resources decisions, a distributed model is expected to give better results than a lumped model, if appropriate information on catchment characteristics can be obtained.

## 5. Optimal Interaction Between Hydrological Models and Remote Sensing

A considerable amount of spatial information on several hydrological variables can now be provided by use of remote sensing techniques. Such data represent new sources of information with potentially large scopes for practical application in hydrology. However, with the exception of snowcover data, remote sensing has generally not been used operationally for hydrological purposes until now. One of the reasons for this gap between the potential and the realized utilization appears to be the uncertainties in interpretation of such data.

On the other hand, it is evident that distributed hydrological models badly need spatial data for calibration and validation purposes. This is particularly true in cases where the aim of the hydrological modeling goes beyond simple predictions of catchment runoff.

Thus, it is realized that both remote sensing data and hydrological model predictions of the same variables, e.g., soil moisture or actual evapotranspiration at a certain grid, will contain uncertainty. Therefore, it is suggested that an optimal way of combining the information from those two sources would be to feed in the remote sensing data to the hydrological model by use of some updating (data assimilation) technique such as Kalman filtering. In this way, the most likely value of the hydrological variables can be found by weighting of the remote sensing and the model estimates by explicitly taking the respective uncertainties in the two methods into account.

It appears evident that such interaction between remote sensing data and distributed hydrological models will prove to be of immense importance for the further developments and applications of both methods.

## 6. Discussion of Requirements to Hydrological Models for Coupling to Global Circulation Models for Climate Change Modeling

In order to ensure and verify accurate predictive capabilities for nonstationary climatic conditions, a hydrological model needs to undergo a more advanced test procedure than the standard split-sample test. In this paper, the hierarchial test scheme, originally proposed by Klemes (1985), has been successfully used.

The present type of hydrological models are, in general, suitable for runoff simulations under nonstationary conditions. This does not mean that all hydrological models are suitable, nor that research and improvements are not required; but rather that it will be possible to identify models which are suitable, either directly or after minor modifications. Whether the same hydrological models are adequate also for simulation of other hydrological variables than runoff under nonstationary conditions has not been addressed in the present paper.

Based on the results presented in this paper as well as the experience of the author, the runoff data requirements for calibration of hydrological models may be summarized as follows:

- In general, 3-5 years of runoff data are sufficient to carry out an accurate calibration. More data are likely to improve the accuracy further, but most often only marginally.

- However, even a calibration against only one year of runoff data is most often sufficient to remove the main part of the prediction uncertainty. This is particularly true in cases where calibration is possible on longer time series from neighbouring catchments with a similar hydrological regime.

Hence, assuming limited availability of resources for collection of runoff data, a combination of long time series from a few stations and short time series from many stations will be optimal.

For simulation of other hydrological variables such as evaporation and energy flux to the atmosphere, it will be necessary to carry out calibration/validation on the same data types. In such a case, it is not sufficient to calibrate against runoff data.

In relation to global change, a coupling between Global Circulation Models (GCMs) and hydrological models is interesting. In this regard, two issues are important, namely the scale of operation and the processes and outputs being described on that scale:

Scale of operation. A GCM usually operates with a resolution which is much larger than the typical resolution of a lumped catchment model (100-5000 km$^2$), which again is much larger than the grid size in a distributed catchment model (0.01-25 km$^2$). Thus, seen from the GCM point of view, the resolution of the output from the lumped hydrological model is sufficient. On the other hand, for application of GCM output in hydrological models, a transformation to a finer resolution is required. This disaggregation must take into account, among others, the effects of the local topography.

Process descriptions and outputs. The outputs from lumped hydrological models of the NAM type are average catchment values of:

- runoff and
- evaporation.

By use of distributed, physically based models, among others, the following outputs may be generated on a grid basis:

- runoff,
- soil moisture,
- surface temperature,
- evaporation, i.e., water flux to the atmosphere,
- energy flux into the soil, and
- energy flux to the atmosphere.

A GCM generates outputs in terms of precipitation, temperaturem and radiation. A GCM, on the other hand, requires information on energy and water flux from the land surface as boundary conditions.

Thus, it is seen that some types of distributed, physically based hydrological models and GCMs can provide each other with the necessary information required for a successful coupling, whereas the traditional lumped, conceptual hydrological model cannot provide outputs with regard to energy flux to the atmosphere.

Global environmental change is an extremely complex issue, where changes may appear successively in different parts of the ecosystems. For instance, with regard to hydrology, a climate change will cause changes in vegetation characteristics: farmers may choose other crops, the time variation of root depths and leaf area index may change, etc. These changes are not straightforward to incorporate in hydrological models. Lumped, conceptual models cannot take such changes into accountat all, whereas it is possible to some extent to consider such aspects in distributed, physically based models.

## 7. Acknowledgements

The modeling work on the Zimbabwe catchments was carried out by my colleagues, Jesper Knudsen, Borge Storm, Merete Styczen, and Roar Jensen. During the data collection and field reconnaissance in Zimbabwe, kind help and assistance were provided by the University of Zimbabwe, National Herbarium, Department of Meteorological Services and Hydrological Branch, Ministry of Energy, and Water Resources and Development.

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# Hydraulic-hydrological simulations of canal-command for irrigation water management

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**Abstract.** A modelling system that combines the hydraulic simulations of the canal and hydrological simulations of the irrigated command is introduced. It uses MIKE 11 and MIKE SHE, two well-established modelling systems, for the hydraulic and hydrological simulations respectively. In addition, it also has an irrigation scheduling module and a crop growth module. The modelling system is applied to the Mahanadi Reservoir Irrigation Scheme, a large irrigation project in Central India. The results show that presently a significant amount of water is wasted in the command during the monsoon season. It is demonstrated that the minimization of this wastage could lead to a substantial crop production in the subsequent dry season. Furthermore, the simulations illustrate the versatility of the modelling system for planning and analysing the various aspects of an irrigation project.

**Key words:** command hydrology, crop growth, India, irrigation management, Madhya Pradesh, modelling

## Introduction

Irrigated agriculture plays a critical role in the economic and social development of the nonindustrialized nations in the world, particularly in South-Asia. Irrigation is directly responsible for complete self-sufficiency or surplus of food production in India, Pakistan and Sri Lanka (Chambers 1988). It has also increased employment opportunities and improved the economic conditions of agricultural labourers (Chitale 1994).

However, in spite of these benefits, the return from the phenomenal investments in large irrigation projects in the region has been disappointing. The major irrigation projects perform at a low overall efficiency of 30–35% (Sanmuganathan & Bolton 1988). It is also estimated that owing to industrial and municipal needs, the percentage share of the water resource to the irrigation sector will decline steadily in the future (Biswas 1994). Thus, in future, irrigation has to become efficient and produce more with less water. This realization has shifted the focus of the policy makers and researchers to the improvement

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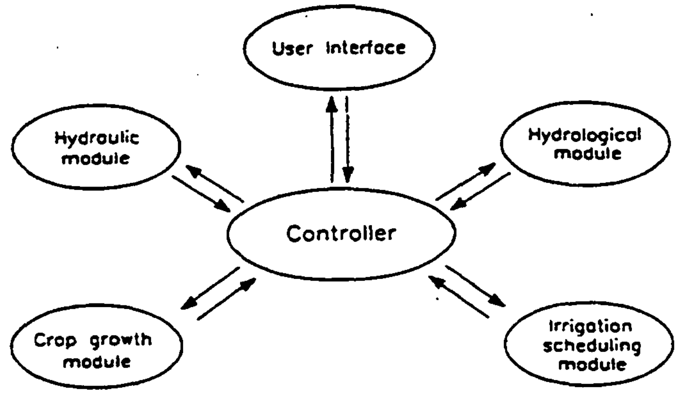


*Figure 1.* Outline of modules and their interrelations in the integrated modelling system.

of canal irrigation performance through main system management (Chambers 1988; Lenton 1994; Loof et al. 1994).

Over the years, several mathematical models have been developed for canal operation and automation (Clemmens & Replogle 1989; Loof et al. 1991; Malaterre 1995). These models are either upstream-control oriented or downstream-control oriented; and employ either local or central criteria to handle scheduled, arranged or on demand methods of water delivery (Reddy 1990; Merkley & Walker 1991). However, these operational models concentrate exclusively on the hydraulic aspect of the canal system and do not take the hydrology of the irrigated command into account. Further, some of these models assume a priori-knowledge of the irrigation demand, though a standard deviation of 30–40% of the mean is possible in seasonal irrigation demand estimates (Mizyed et al. 1991).

In the present paper, a modelling system is presented in which the hydraulic simulations of the canal system and the hydrological simulations of the irrigated command are conducted simultaneously. The model is applied to a major canal irrigation project in India and its advantages in efficient irrigation planning are presented.

## Description of the modelling system

The developed modelling system has a modular structure (Fig. 1). The core of the system is the *Controller module*, which controls and steers data flow among various modules. The details of the different modules are as follows.

### *Hydraulic module*

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The transport of water through the canal system is modelled using the *hydrodynamic module (HD)* of the *MIKE 11*, the one-dimensional river simulation

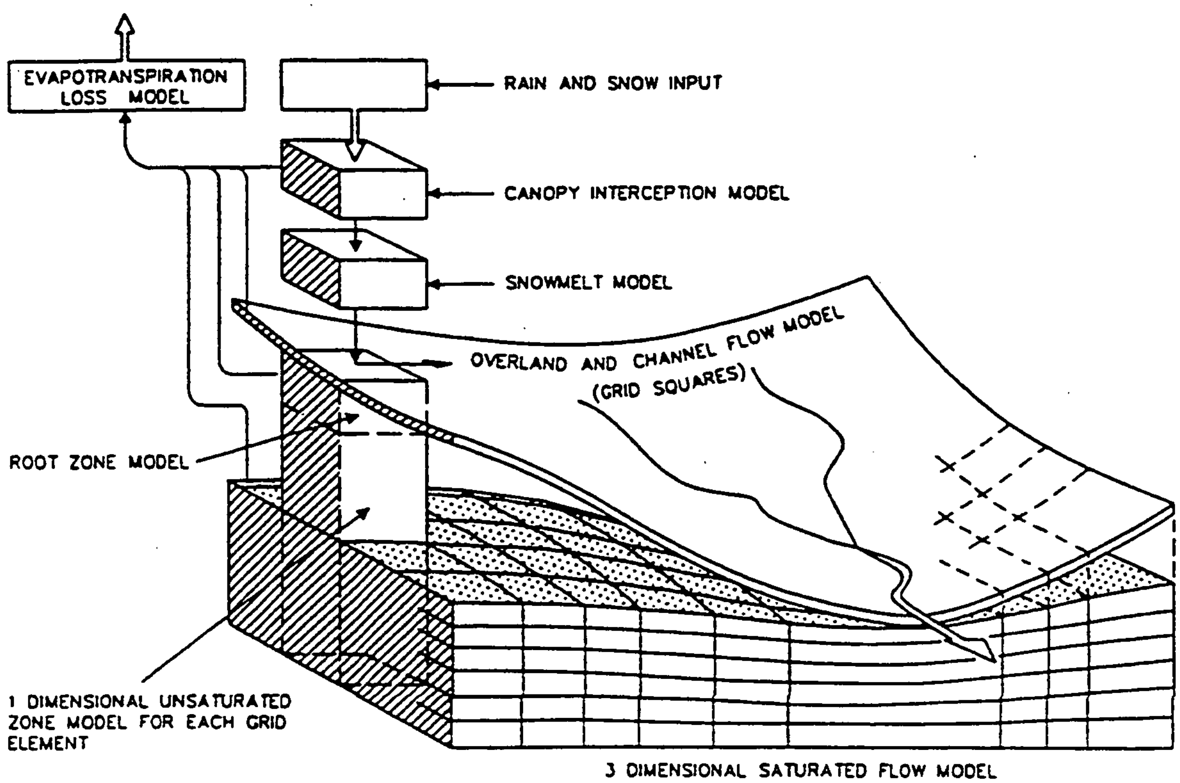


*Figure 2.* Schematic representation of the MIKE SHE model structure.

modelling system (Havnø et al. 1995). *MIKE 11 HD* solves the Saint-Venant equations, transformed to a set of implicit finite difference equations, using double sweep algorithm (Abbott & Ionescu 1967; Abbott 1979). It includes the description of flow over a variety of hydraulic structures, generally encountered in an irrigation system. It also includes the possibility of simulating the operation of gates or head regulators in canals.

*Hydrological module*

The water movement in the irrigated command is modelled by *MIKE SHE*, a generalized mathematical modelling system capable of describing the entire land phase of the hydrological cycle in a given command. The model area is discretized by two analogous horizontal-grid square networks for surface and ground water flow components. These are linked by a vertical column of nodes at each grid representing the unsaturated zone (Fig. 2). A finite difference solution of the partial differential equations, describing the processes of overland and channel flow, unsaturated and saturated flow, interception and evapotranspiration, is used for water movement modelling. A brief description of the components of *MIKE SHE* is given in the following. For a more detailed description, see Abbott et al. (1986 a, b) and Refsgaard and Storm (1995).

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*Interception and evapotranspiration component*

The interception process is modelled by introducing an interception storage, expressed as a function of leaf area index (Jensen 1983). The actual evapotranspiration is calculated based on the potential evapotranspiration using the Kristensen and Jensen model (Kristensen & Jensen 1975). Here, the actual evapotranspiration rate is further adjusted according to vegetation density and water content in the root zone.

*Overland and channel flow component*

The overland flow process is simulated in each grid square by solving the two-dimensional diffusive wave approximation of the Saint-Venant equations. For the river drainage network channel flow is calculated on the basis of the one-dimensional form of the equation which is solved in a separate node system located along boundaries of the grid squares. However, as MIKE SHE's channel flow component can not handle the effects of hydraulic constrol structures, the calculation of channel water levels and flows in the present integrated modelling system is instead taken care of by the hydrodynamic module of MIKE 11 described above.

*Unsaturated zone component*

Soil moisture distribution in the unsaturated zone is calculated by solving the one-dimensional Richards' equation. Extraction of moisture for transpiration and soil evaporation is introduced via sink terms at the node points in the root zone. Infiltration rates are found by the upper boundary that may be either flux controlled or head controlled. The lowest node point included 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 component*

The ground water flow is modelled using an implicit finite difference solution of the two-dimensional nonlinear Boussinesq equation for an unconfined aquifer. The interaction between the streamflow and groundwater systems is calculated on the basis of water levels in the river system and the ground water tables.

*Irrigation scheduling module*

The irrigation scheduling module is based on water balance technique and uses either the soil moisture approach or the water level approach. In the soil moisture approach, the irrigation demand is initiated when the soil moisture in the root zone reaches a critical value, defined in terms of available soil water

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and maximum allowable depletion (MAD). In the water level approach, used exclusively for paddy (except during ripening and grain formation stages), irrigation demand is initiated when the water level on the field surface reaches a lower limit. The water levels are defined as a function of crop growth stage. These approaches are similar to those proposed by Singh et al. (1995) and Azhar et al. (1992), respectively. However, here MIKE SHE does the water balance calculations, based on which the irrigation scheduling module calculates the irrigation demand.

*Crop growth module*

The crop growth module involves dynamic modelling of leaf area index and yield, i.e. dry matter production, based on phenological parameters describing the crop development (Thornley 1976; Hansen et al. 1993). The governing equations are as follows.

$$\frac{dY}{dt} = RYe^{-st} \tag{1}$$

$$LAI = \begin{cases} Y(SLA) & for\ t < t_{max} \\ Y(SLA)(1 - \frac{a(t-t_{max})}{(t-t_{max})+D}) & for\ t \geq t_{max} \end{cases} \tag{2}$$

$$RD = \begin{cases} \frac{RD_{max}t}{t_{max}} & for\ t < t_{max} \\ RD_{max} & for\ t \geq t_{max} \end{cases} \tag{3}$$

where Y = dry matter production, g/m$^2$; R = initial relative growth rate; S = senescence parameter; t = time (days after crop emergence); SLA = specific leaf area, m$^2$/g; $t_{max}$ = time to maximum LAI, days; a, D = leaf area damping parameters, RD = root depth, m; and $RD_{max}$ = maximum root depth, m.

Daily potential and actual yields, leaf area index and yield loss due to moisture stress are the main outputs from this module. The yield loss due to water stress follows the FAO relationship (Doorenbos & Kassam 1979), modified by replacing evapotranspiration terms with transpiration. The modified relationship is as follows.

$$(1 - \frac{Y_a}{Y_m}) = \sum_{i=1}^{n} K_y^i (1 - \frac{E_{at}}{E_{mt}}) \tag{4}$$

where $Y_a$, $Y_m$ = actual and maximum yields; $E_{at}$ = actual transpiration, $E_{mt}$ = maximum transpiration that would have occurred if there had been no water stress; $K_y$ = yield response factor; and i = crop growth stage.

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Equation (4) performs better than the original FAO relationship when used with the Kristensen and Jensen model (Jørgensen 1995). $E_{at}$ and $E_{mt}$ in Eq. (4) are estimated by deducting the soil evaporation from the actual and maximum evapotranspiration values, where the soil evaporation and actual evapotranspiration are estimated using the Kristensen-Jensen model.

Since parameters R, S, SLA, a and D in Eqs. (1) and (2) are not readily available, a provision is made to fit the growth curves according to the user specified maximum LAI, time to maximum LAI and maximum root depth.

*Coupling of modules in the modelling system*

The different modules in the modelling system operate interactively. For example, the hydraulic module receives the irrigation demand from the controller, while it sends the information about the hydrodynamic state of the canal system to the controller that transmits it to the hydrological module. The hydrological module then performs the water balance calculations and sends information about the state of the individual fields in terms of potential and actual evapotranspiration, effective and maximum moisture content integrated over the root depth, recharge to the ground water and depth of the ground water table to the controller module. The controller module then transmits this information to the irrigation scheduling and crop growth modules. Based on the information received, the irrigation scheduling module calculates the irrigation demand and sends it to the controller module, whereas the crop growth module calculates the crop yield, and so on.

## Case study

The modelling system is applied to the Mahanadi Reservoir Irrigation Scheme (MRP), situated in the southeastern part of Madhya Pradesh, India, between latitude 20° and 22° N and longitude 81° to 83° E. Data on various aspects of the command area viz. topography, geology, soils, crops, main canal system and daily canal releases during 1991–1993 are obtained from the Irrigation Department, MRP.

*Project description*

The MRP complex consists of six interlinked reservoirs (Fig. 3). It is designed to irrigate an area of 374 000 ha in kharif (monsoon) season (July-October) and 131 000 ha in rabi (winter) season (November-February), through five interlinked canal systems, besides meeting the municipal and industrial demands in the adjoining area. However, in the present study, only the Mahanadi main

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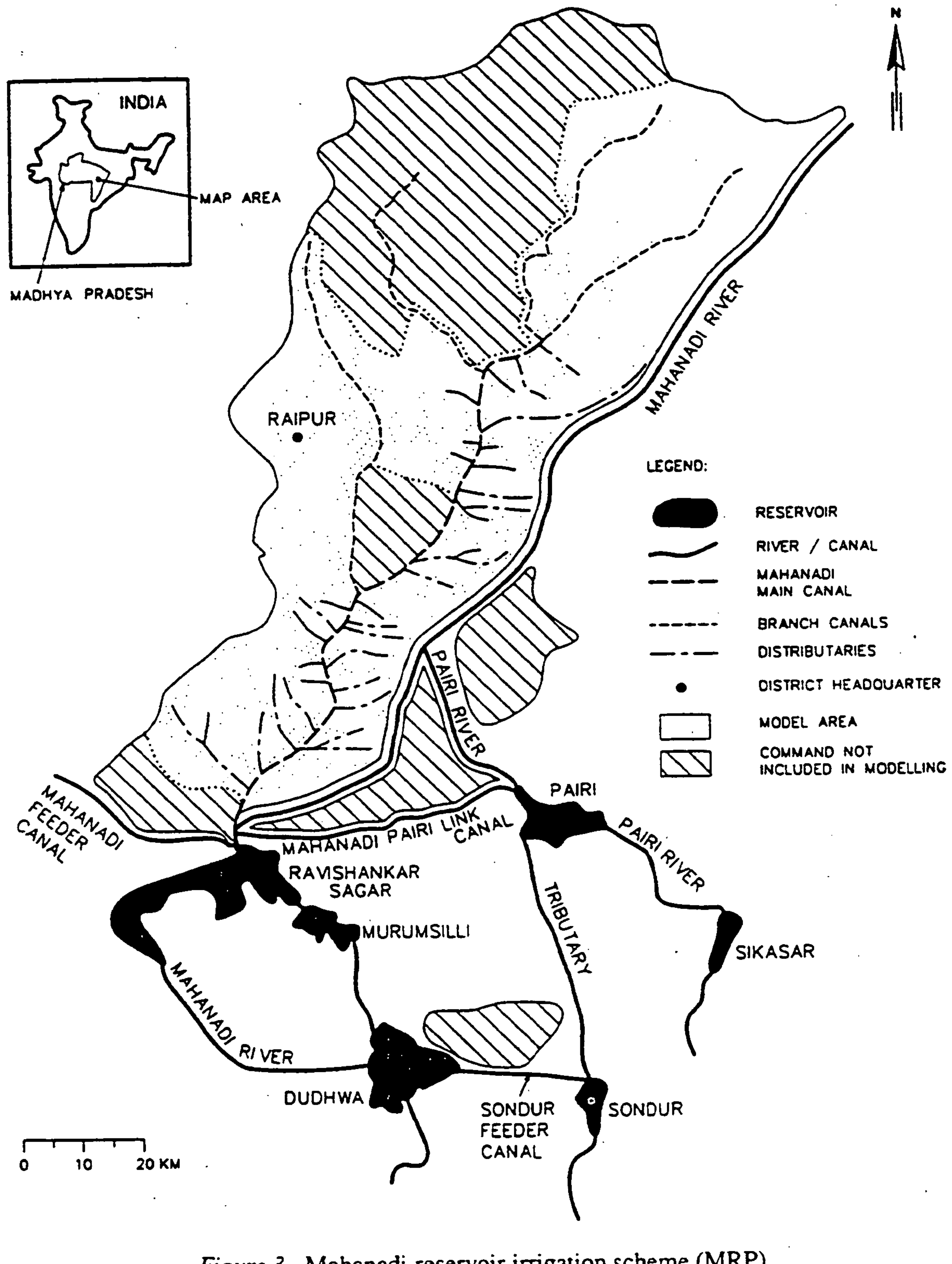


*Figure 3.* Mahanadi reservoir irrigation scheme (MRP).

canal command, which accounts for 197 460 ha of the design area, is used for modelling.

The climate is subtropical with three well-defined seasons. The average annual rainfall recorded at Raipur is 1100 mm, 90% of which occurs during southwest monsoon (June to September). The topography is predominantly

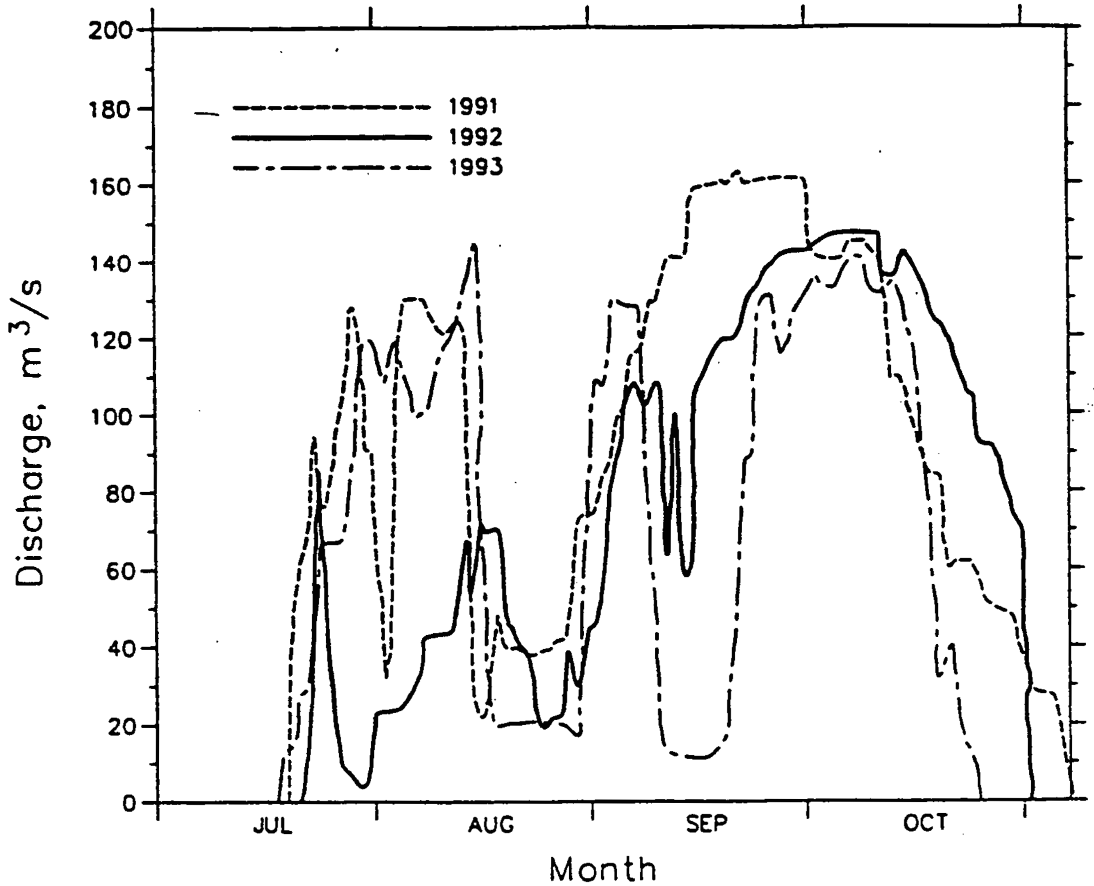


*Figure 4.* Mahanadi main canal releases during 1991–1993.

flat, with 50% of the command having less than 1% slope. The major agricultural crops are paddy during kharif season and wheat, pulses and vegetables during rabi season.

The soils in the command are broadly classified as heavy (clay loam) and light (sandy loam) covering 71% and 29% of the area, respectively.

*Main canal system*

The Mahanadi main canal is 116 km long with a design discharge capacity of 391 $m^3/s$ at the head end. It supplies water to four branch canals, several distributaries and field outlets. The main and branch canals are lined. The main canal, branch canals and distributaries are each equipped with a head regulator. In addition, four cross regulators are provided along the main canal.

Figure 4 presents the daily releases to the main canal during 1991–1993. It is evident from the figure that the irrigation releases are presently limited to only kharif season.

*Setup preparation*

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*Hydraulic module*

The setup preparation for the hydraulic module involves specifications of canal cross-sections, head and cross regulators, upstream and downstream boundary conditions and seepage losses in the MIKE 11 HD.

To reduce the computational requirements, the canal system is simplified here. Consequently, the main canal is included in the model setup whereas the branch canals and distributaries are merged to 16 distribution channels. The function of these distribution channels is to direct the correct amount of water from the main canal to the command. This is done by calculating the release from the main canal through the head regulators at the upstream end of the distribution channels. A constant head boundary condition is used at the downstream end of these distribution channels. This implies that the unsteady flow conditions are not modelled in these channels. For seepage loss in canal system, a value of 0.4 $m^3/s/Mm^2$ of water surface area is used against Government of India recommendation of 0.3 $m^3/s/Mm^2$ of wetted perimeter (Ministry of Irrigation 1984).

*Hydrological module*

The setup preparation for the hydrological module involves specifications of topography, geology, soil distribution, meteorological characteristics, soil physical characteristics and vegetation data for the command.

The setup here is also simplified to represent 64 fields, i.e. each distribution channel is linked to four fields, two each for heavy and light soils. Fig. 5 presents the modified setup for hydraulic-hydrological module linkage.

Daily data on rainfall and pan evaporation for three years, i.e. 1991–1993, are obtained for a climatological station in Raipur from the Meteorological Department, Indira Gandhi Agricultural University, Raipur. Fig. 6 presents the daily rainfall and pan evaporation data.

Data on soil physical characteristics viz. field capacity, wilting point and soil water retention curve are determined from the literature (Katre 1992). The saturated hydraulic conductivities are estimated from double-ring infiltrometer test results (Agrawal 1994). Table 1 presents the physical characteristics of soils in the command. The crops considered here are paddy, transplanted and direct sown, in kharif season and wheat, mustard, gram and potato in rabi season. Data on areal distribution, leaf area index and maximum root depth of crops is obtained from the Irrigation Department, MRP and Agronomy Department, Indira Gandhi Agricultural University, Raipur. Crop coefficients, $K_c$, for Indian conditions, and yield response factor, $K_y$, are determined from literature (Mazumdar 1983; Doorenbos & Kassam 1979). Table 2 summarizes the relevant information.

*Irrigation scheduling module*

In this module, the maximum allowable depletion (MAD) values in the soil

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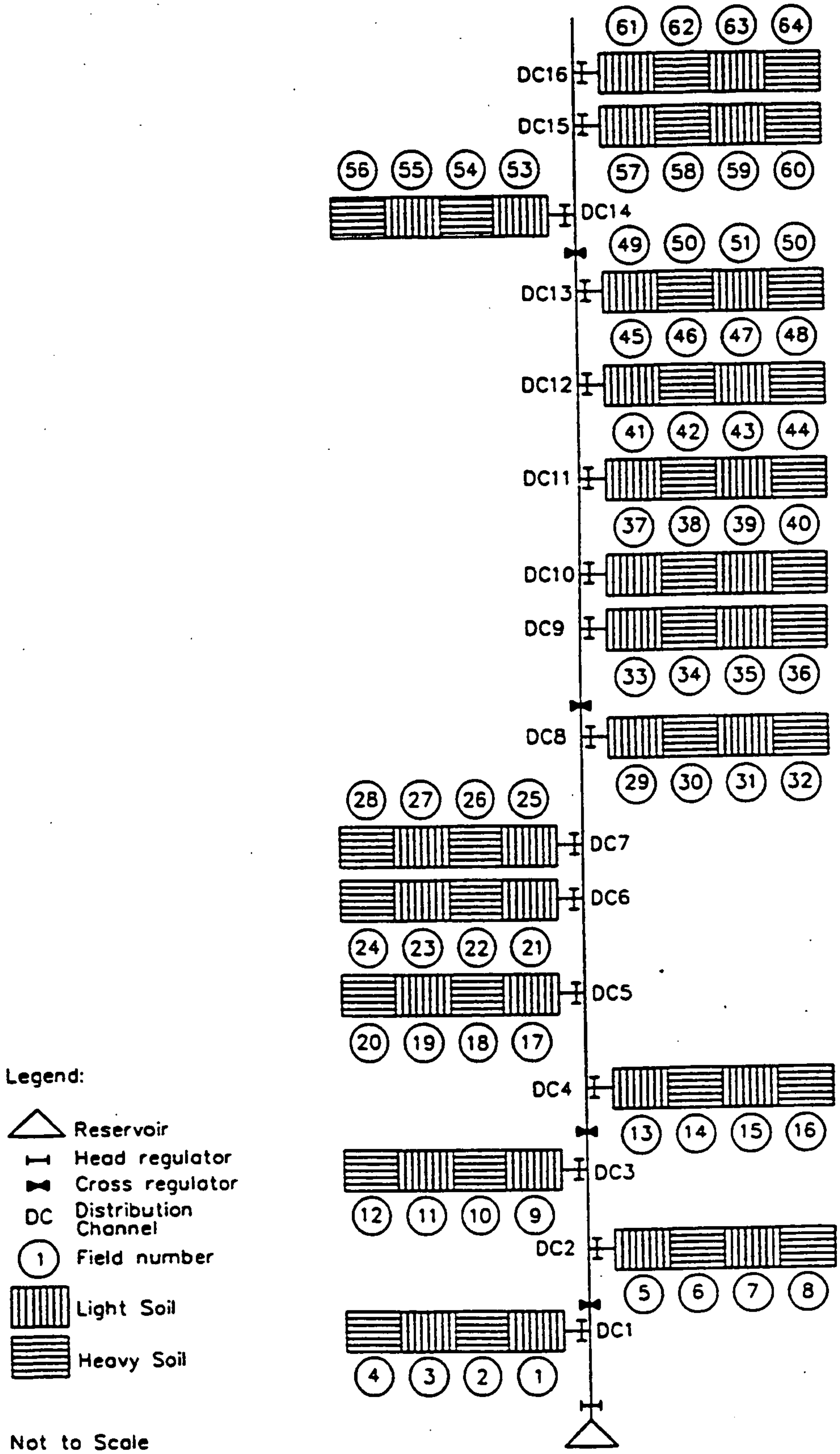


*Figure 5.* Schematic hydraulic-hydrological model setup for the MRP command area.

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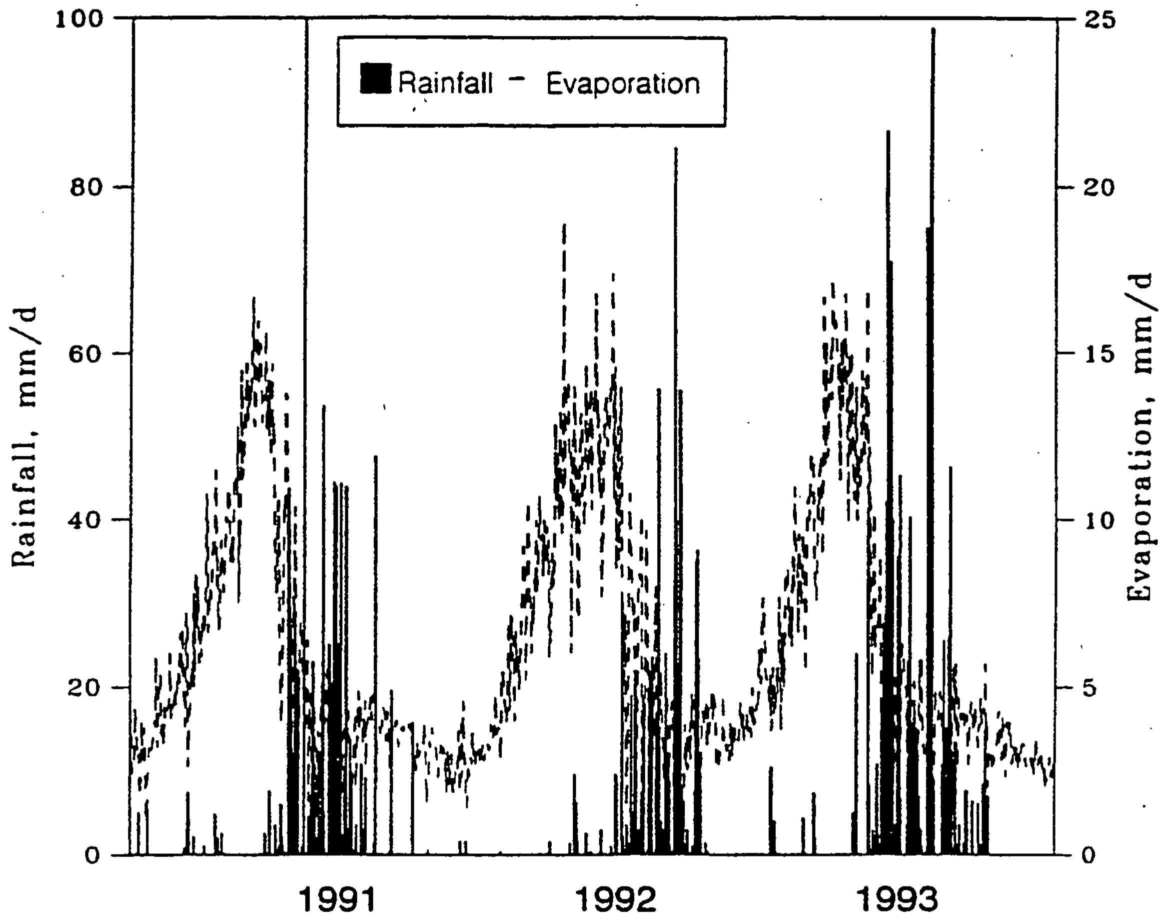


*Figure 6.* Rainfall and evaporation data.

*Table 1.* Physical characteristics of the MRP soils.

| Soil | Texture | | | Field capacity | Wilting point | Saturated hydraulic conductivity |
|---|---|---|---|---|---|---|
| | Sand | Silt | Clay | | | |
| | % | % | % | Vol. % | Vol. % | 7 m/s |
| Light | 40.5 | 31.5 | 27.0 | 29.1 | 9.8 | $1.0 \times 10^{-5}$ |
| Heavy | 32.5 | 27.5 | 40.0 | 34.2 | 16.7 | $7.0 \times 10^{-6}$ |

moisture approach and water levels on field surface in the water level approach are specified. In the soil moisture approach, MAD value of 0.50 is used (Stegman 1983). In the water level approach, the water levels recommended by Doorenbos and Kassam (1979) at different growth stages of paddy are used.

The irrigation scheduling module works for both the 'rotational' and the 'on demand' schedules (Case I and II described below). However, for a rotational schedule its application is limited to supplying irrigation water to the fields during periods where water is available in the local distribution channel

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*Table 2.* Important features of the selected crops.

| Information | Crop | | | | | |
|---|---|---|---|---|---|---|
| | Paddy (T)* | Paddy (DS)** | Wheat | Gram | Mustard | Potato |
| Duration (days) | 120 | 120 | 105 | 100 | 100 | 120 |
| Sowing period (week no.) | 26–28 | 26–28 | 45–47 | 46–47 | 46–47 | 46–47 |
| Maximum yield (ton/ha) | 3.5 | 3.0 | 2.0 | 1.0 | 1.2 | 12.0 |
| $K_c$ | | | | | | |
| –Establishment | 1.10 | 1.10 | 1.15 | 0.15 | 0.36 | 0.36 |
| –Vegetative | 1.10 | 1.10 | 0.52 | 0.52 | 0.97 | 0.97 |
| –Flowering | 1.10 | 1.10 | 0.88 | 0.88 | 0.99 | 0.99 |
| –Yield formation | 0.95 | 0.95 | 0.70 | 0.70 | 0.46 | 0.46 |
| –Ripening | 0.95 | 0.95 | 0.20 | 0.20 | 0.20 | 0.20 |
| $K_y$ | | | | | | |
| –Establishment | 1.75 | 1.75 | 0.20 | 0.20 | 0.20 | 0.45 |
| –Vegetative | 1.75 | 1.75 | 0.20 | 0.20 | 0.20 | 0.45 |
| –Flowering | 2.25 | 2.25 | 0.60 | 0.65 | 0.50 | 0.70 |
| –Yield formation and ripening | 0.30 | 0.30 | 0.30 | 0.30 | 0.25 | 0.20 |

*Transplanted
**Direct sown

and the field at the same time requires water according to the above criteria. For a on demand schedule the module also activates the head regulator at the distribution channel enabling water to flow from the main canal to the distribution channel.

*Crop growth module*

In this module, senescence parameter, relative growth rate, maximum expected yield, specific leaf area index, time to maximum leaf area index, leaf area damping parameters, crop coefficients and yield response factor at different growth stages are specified for each crop. Here the maximum LAI, time to maximum LAI and maximum root depth are used to fit the crop growth functions.

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### *Model simulations and results*

#### *Kharif simulations*

Simulations are made for the 1991, 1992 and 1993 kharif seasons over a period of fourteen weeks. The time interval used in the crop growth and irrigation scheduling modules is one day, whereas finer time steps (down to hours) are used in the hydraulic routing and soil moisture calculations when required for numerical reasons. During the kharif season the entire command is considered for cultivation with transplanted and direct sown paddies occupying 28.2% and 71.8% of the area respectively. For irrigation, the following two scheduling practices are considered.

Case I: Here the weekly rotational schedule of distribution channels, a general practice in the command area, is adopted. In this practice, the irrigation schedules are prepared in the beginning of the cropping season implying that the periods for which water is released from the main canal to the different distribution channels are predetermined and fixed. Thus, the crop water requirement for the individual fields, which in the integrated model is calculated by the irrigation scheduling module may or may not be met, depending on the rotational schedule of the corresponding distribution channel.

Case II: Here "on demand" irrigation is considered, subject to the water level state of the main canal system (determined by the St. Venant Equations). Here also the crop water demand for the individual fields is calculated by the irrigation scheduling module. However, contrary to Case I, here the cumulative crop water demand of a particular distribution channel is immediately met if the water is available in the main canal, so that the water can be diverted from the main canal to the distribution channel through the head regulator.

However, both in Cases I and II, the present releases from the reservoir to the head of the main canal decided by the management (Fig. 4) are used. Furthermore, as the four crossregulator structures located on the main canal are not operated in practise, they are also kept constant in the model.

Further, the following assumptions are made here:

1. Water is the only limiting factor for the crop growth.
2. Crops are transplanted or sown on the same day all over the command.
3. Water is available to meet the nursery requirements of transplanted paddy.

Table 3 summarizes the simulation results whereas Figs. 7a,b to 9a,b present the typical results from hydrological and crop growth modules for 1991, 1992 and 1993 respectively. In these figures the irrigation and rainfall amounts are shown together with the calculated actual evapotranspiration and water content in the upper meter of the soil. Furthermore the leaf area index, root depth and crop yield as calculated by the crop growth module are presented.

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*Table 3*. Summary of kharif simulation results for two alternative irrigations.

| Simulation type | Potential production million tons | Actual production million tons | Rainfall mm | Actual evapo-transpiratiom mm | Irrigation mm | Canal release $Mm^3$ | Tail loss $Mm^3$ |
|---|---|---|---|---|---|---|---|
| Case I* 1991 | 0.621 | 0.612 (99%)** | 766 | 355 | 215 | 925.92 | 490.54 (53%)!! |
| Case II! 1991 | 00.621 | 0.614 (99%) | 766 | 358 | 238 | 925.92 | 448.69 (49%) |
| Case I 1992 | 0.621 | 0.558 (90%) | 609 | 372 | 169 | 746.65 | 400.78 (54%) |
| Case II 1992 | 0.621 | 0.593 (96%) | 609 | 383 | 176 | 746.65 | 399.71 (54%) |
| Case I 1993 | 0.621 | 0.593 (96%) | 712 | 404 | 169 | 716.06 | 374/16 (52%) |
| Case II 1993 | 0.621 | 0.617 (99%) | 712 | 415 | 185 | 716.06 | 346.12 (48%) |

*Weekly rotation
!On demand
**Values in parentheses show% of potential production
!!Values in parentheses show% of canal release

It is seen that the actual productions are always, except in case I of 1992, over 95% of the potential production (Table 3). This shows the availability of sufficient amount of water during the kharif season.

The rainfall and applied irrigation amounts are highest in 1991. This allows a favourable moisture regime in the unsaturated zone throughout the growing season (Figs 7a and 7b) and results in 99% of potential production in both cases I and II. On the other hand, the rainfall and applied irrigation amounts are lowest in 1992. This results in moisture stress from mid-September onwards in case I (Fig. 8a) and in the beginning of October in case II (Fig. 8b). Since from mid-October the irrigation scheduling approach changes from water level to soil moisture, the effect of moisture stress is more prominent in case I, where the actual production is lowest at 90% of the potential. In 1993, the rainfall and applied irrigation magnitudes are between 1991 and 1992 values and result in actual productions of 96% and 99% of the potential in case I and II respectively. It is interesting to note that case II of 1993 results in highest production. This is because of uniform distribution of rainfall over the growing season, particularly in October (Fig. 9b).

On demand irrigation (Figs. 7b, 8b and 9b) results in uniform moisture distribution in the unsaturated zone compared to rotational schedule (Figs. 7a, 8a and 9a) and higher production, as expected. Its effects on moisture

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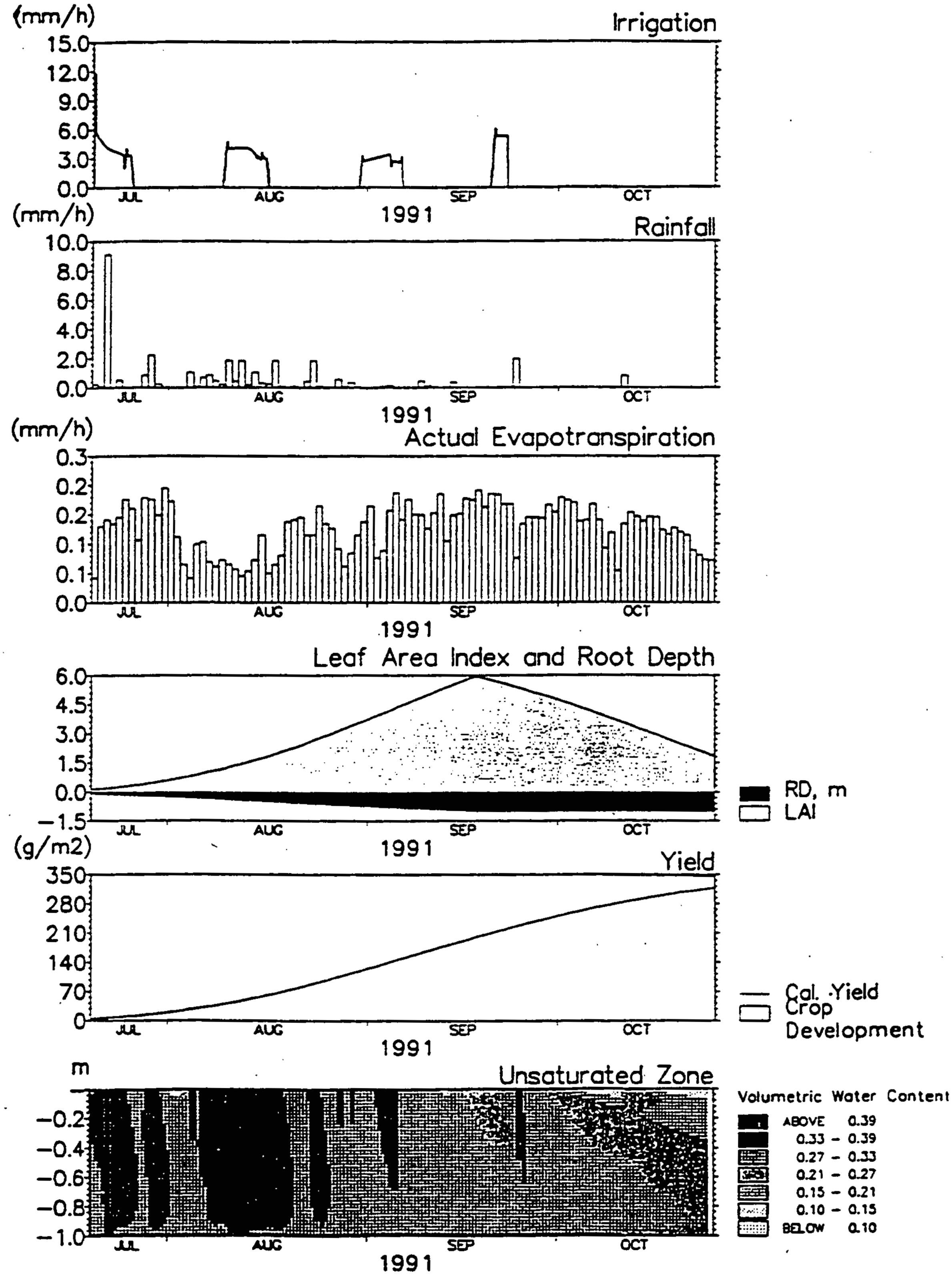


*Figure 7a.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1991- Case I: rotational irrigation schedule.

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regime in unsaturated zone and actual production are more prominent in 1992 and 1993 when canal releases are low. Though it may not be feasible to

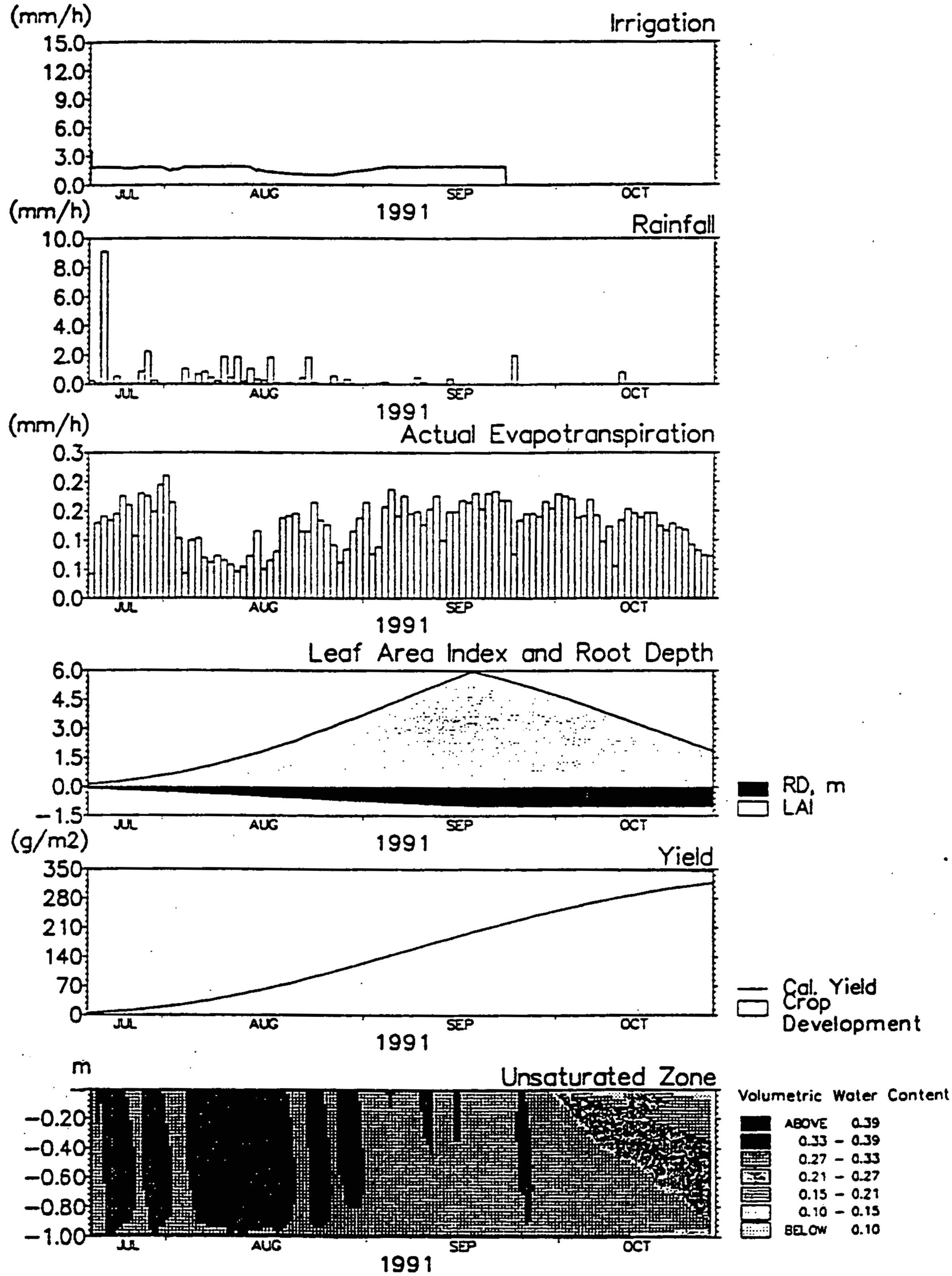


*Figure 7b.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1991- Case II: on demand irrigation schedule.

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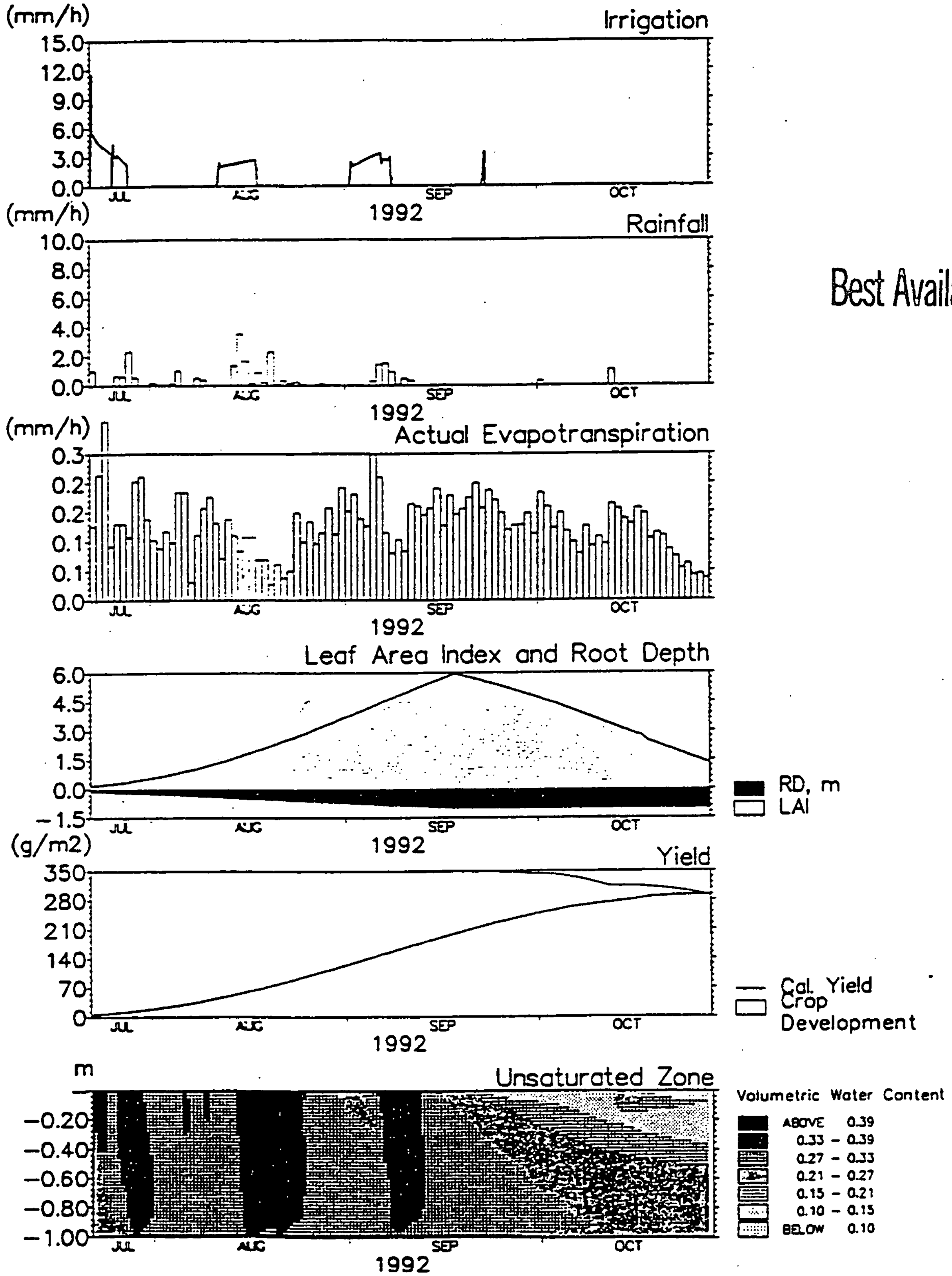




*Figure 8a.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1992- Case I: rotational irrigation schedule.

adopt the on demand irrigation in the field conditions presently due to lack of infrastructure in the command, it may serve as a guideline for scheduling the

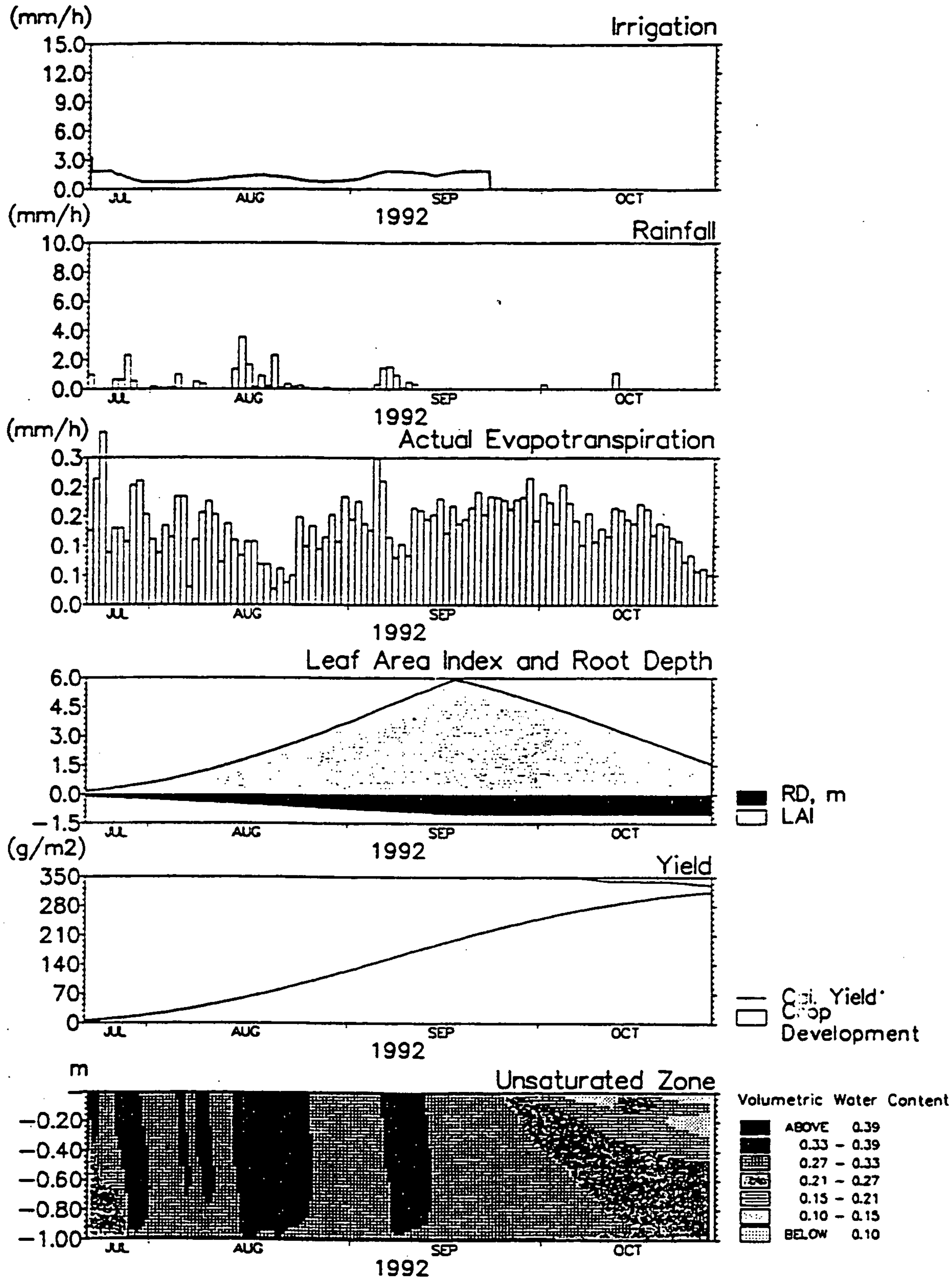


*Figure 8b.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1992- Case II: on demand irrigation schedule.

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rotation of distribution channels to enhance the water use efficiency. In the present simulations on demand irrigation scheduling results in daily irrigation,

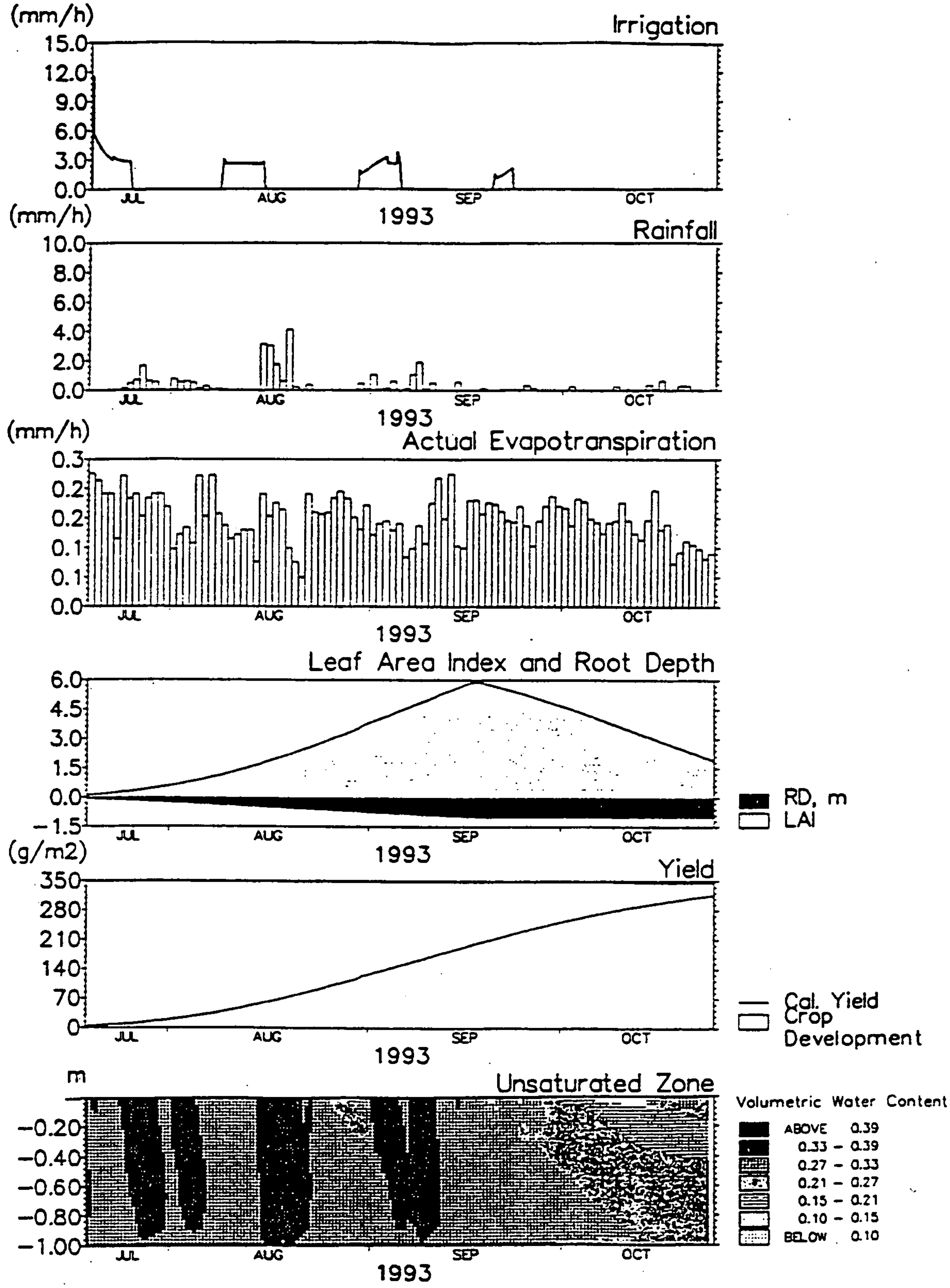


*Figure 9a.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1993- Case I: rotational irrigation schedule.

see the upper parts of Figs 7b, 8b and 9b. This might not be practical with a surface irrigation system as used in this area. However, the results in terms of moisture regime, yield, irrigation amounts and tail end loss would be almost

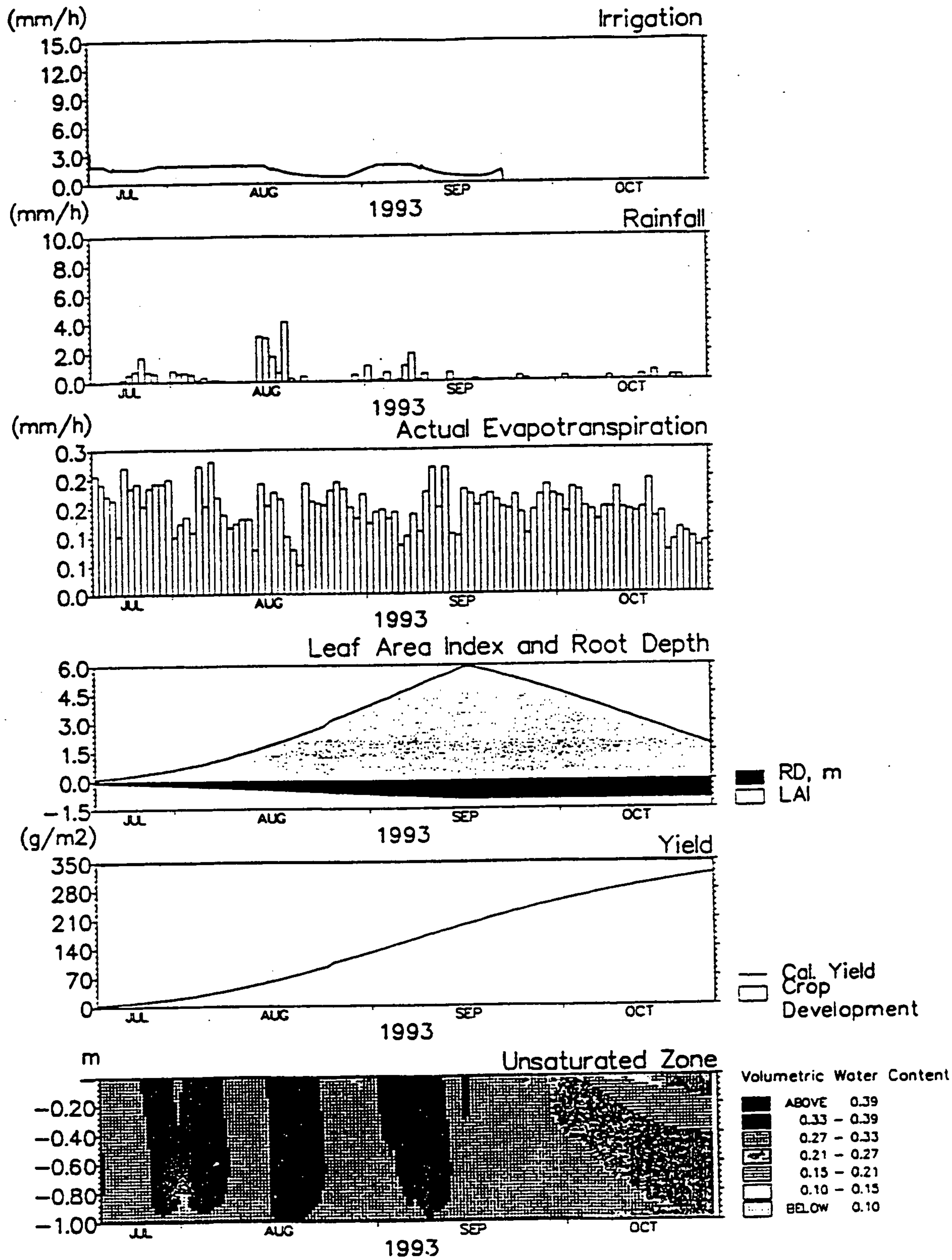


*Figure 9b.* Results from hydrological and crop growth modules for one of the 64 fields for the kharif season of 1993- Case II: on demand irrigation schedule.

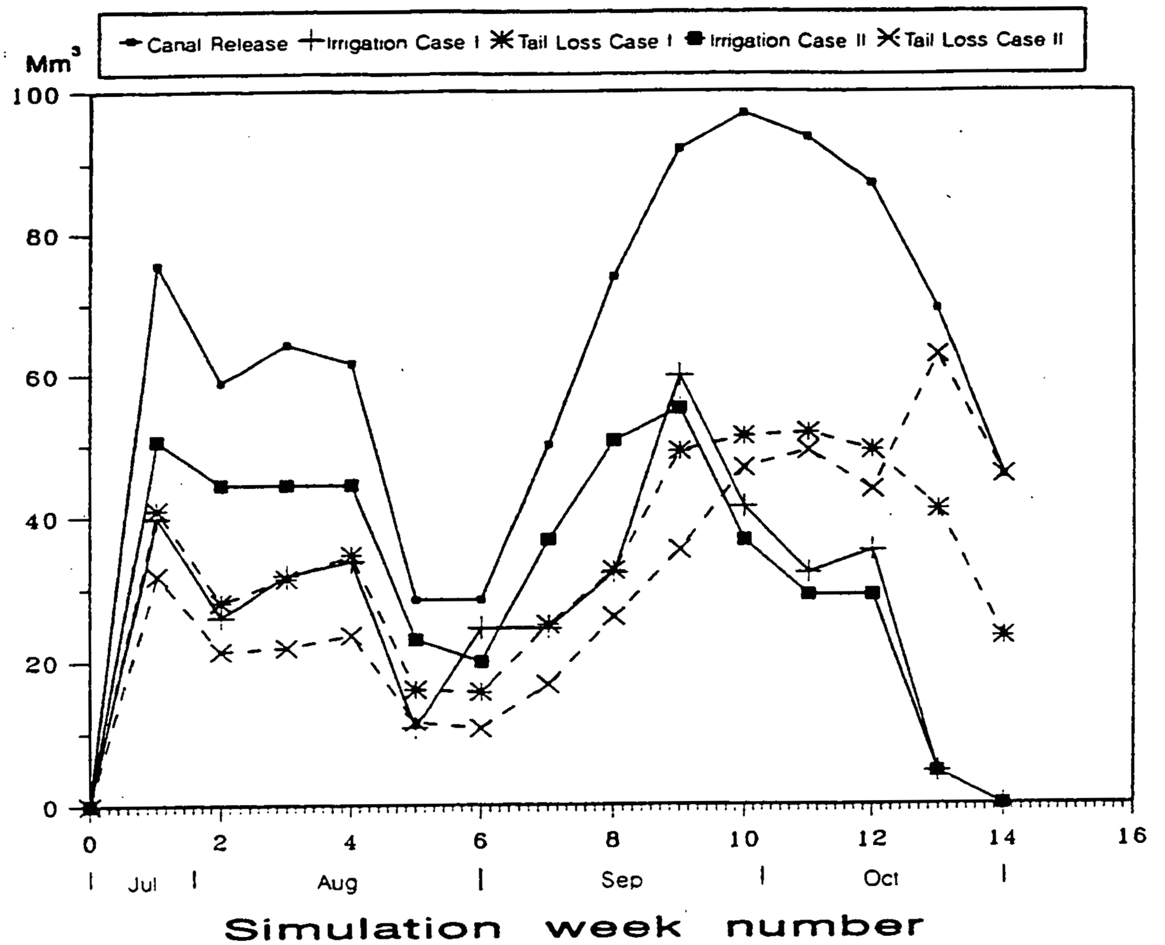


*Figure 10.* Weekly canal release, tail loss and irrigation water utilization for 1991 (kharif season).

identical if the surface irrigation to the individual fields was supplied less frequent, for instance every five day, which still is much more frequent than under the rotational scheme.

The tail losses, as indicated in Table 3, lie between 48% (case II, 1993) and 54% (case I, 1992) of the present canal releases. Though a limited amount of tail loss is unavoidable due to necessity of maintaining a minimum head of water in the main canal to supply all distribution channels, its high value in the simulations shows that the management decisions on fixing the canal releases are presently inefficient. This results in wastage of precious water during kharif season and explains the inability of the management to supply water during rabi season in the command. To illustrate this further, the canal releases, irrigation water utilization and tail losses are presented on weekly basis (Figs. 10 to 12). It is seen that in case I of all three years, the tail losses are close to or, at times, even higher than irrigation. In cases II, however, the tail losses are considerably lower than the irrigation, which shows the better utilization of canal release towards irrigation in this case.

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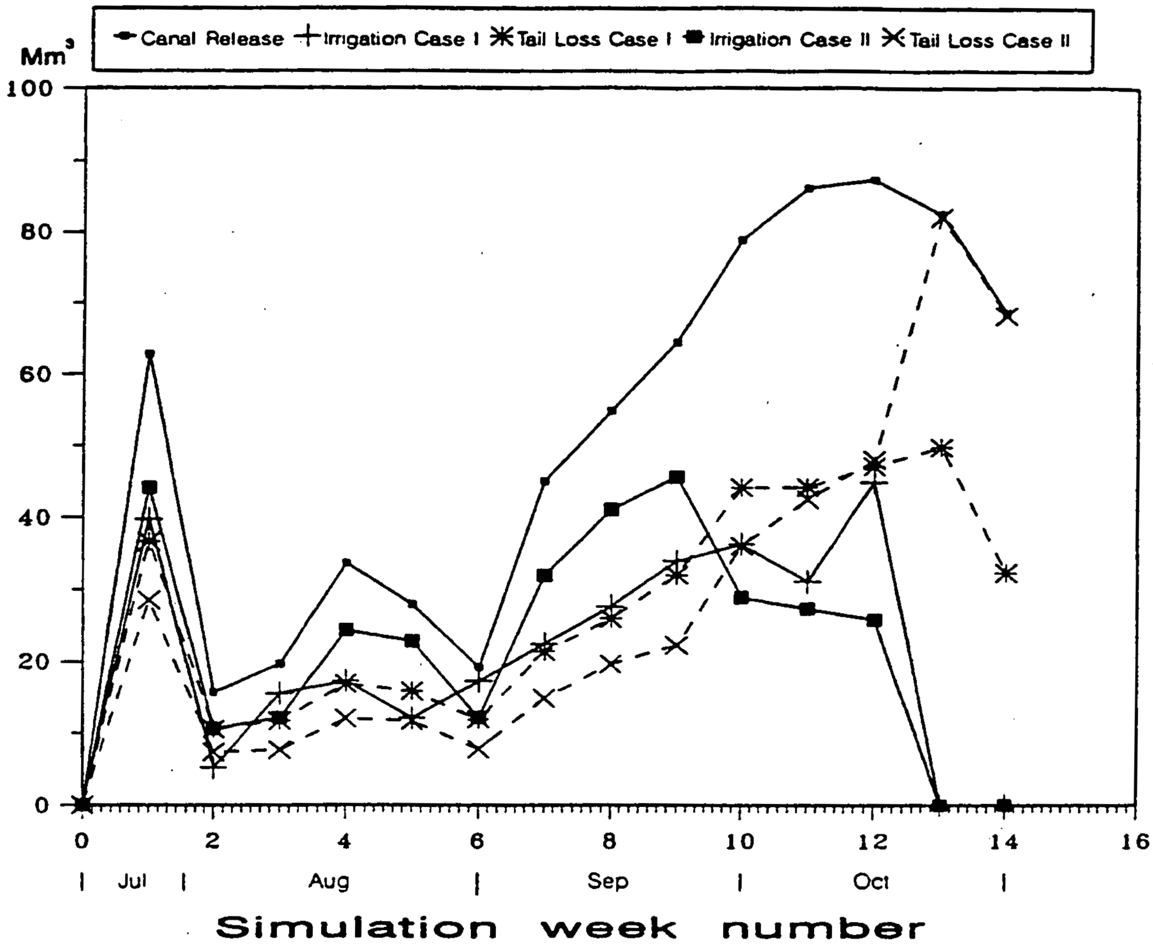


*Figure 11.* Weekly canal release, tail loss and irrigation water utilization for 1992 (kharif season).

It should be emphasized that the amount of tail end loss to a large extent is determined by the release from the reservoir to the main canal for which the actual figures (Fig. 4) have been used in both cases. As this canal release turns out to be much higher than the field irrigation requirements during the wet kharif (monsoon) season the tail end loss becomes very high as compared to the differences between the two scheduling practices. Although the difference between the tail end loss figures in Table 3 therefore may appear small, the results consistently show that water, seen from an irrigation point of view, is more efficiently used with the on demand irrigation scheduling.

It may be noted that irrespective of irrigation criteria, the tail losses are significantly higher in weeks 13 and 14 of all years. An analysis of these two weeks shows that only 4% and 8% of water released is used for irrigation in 1991 and 1993 respectively, whereas in 1992 irrigation utilization is nil. This amounts to a supply of 120 $Mm^3$ over the irrigation requirement during these two weeks. This water could easily be saved and used for rabi irrigation as illustrated later.

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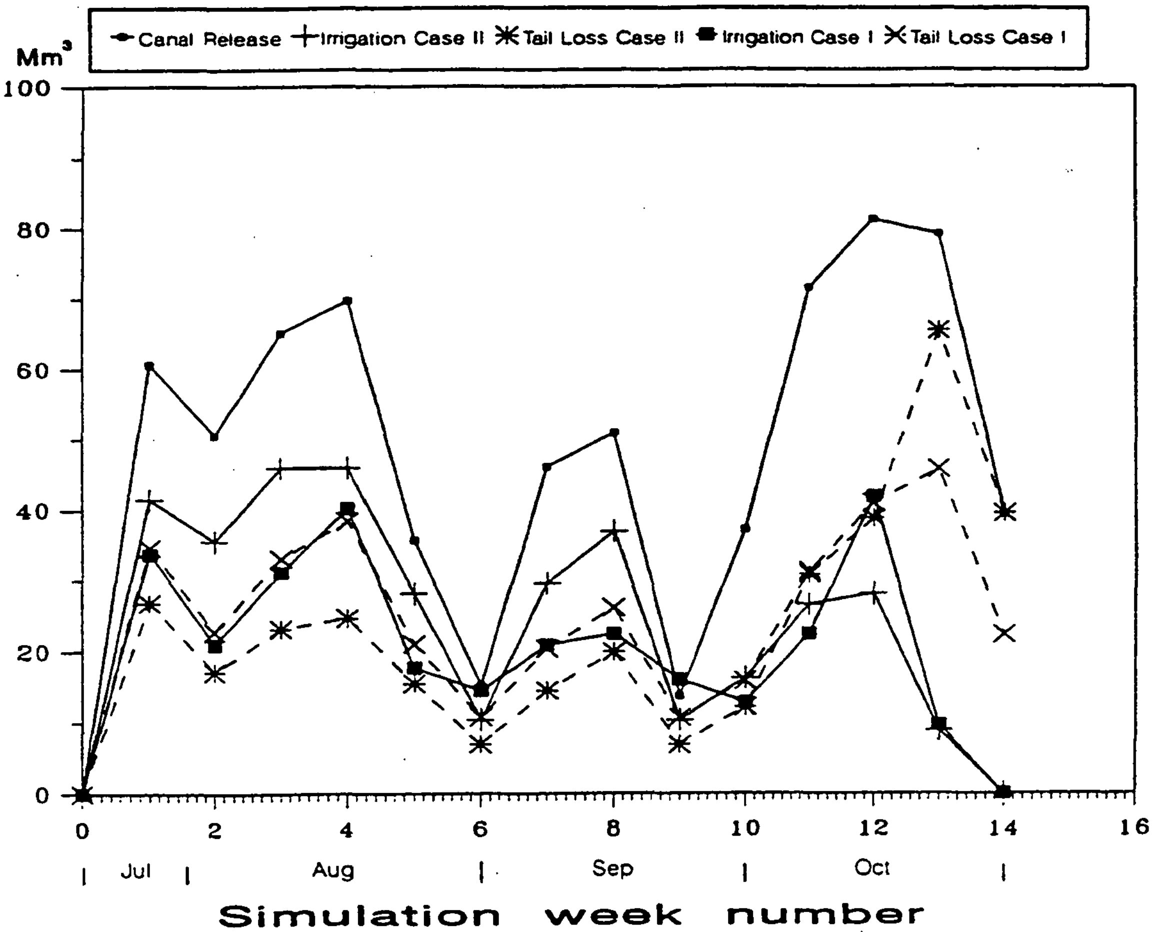


*Figure 12.* Weekly canal release, tail loss and irrigation water utilization for 1993 (kharif season).

*Rabi simulation.*
A simulation of twelve week duration is made for the dry rabi season with 120 Mm$^3$ of canal release. Since the water is not sufficient to irrigate the entire command, according to the MRP design, only 35% of area is considered for cultivation. Further, the canal releases are limited to one week at the time of sowing and three weeks during late vegetative and flowering stages with on demand irrigation. This is because these periods are critical for all crops sown during rabi season. According to the MRP design, 40%, 40%, 12% and 8% of the cultivated area are assigned to wheat, gram, mustard and potato respectively. Further, the crops are distributed at random over the 64 fields.

For the rabi simulations initial soil moisture conditions extracted from model simulations at the end of the kharif season have been used. Although the kharif simulations have all been carried out with the full canal release, without saving the 120 Mm$^3$, and therefore not fully corresponds to a situation where the canal release is reduced with this amount of water, this has no significant impact on the soil moisture conditions, because about 95% of the 120 Mm$^3$ as described above can be saved from the tail end losses during the last two weeks

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*Table 4.* Summary of rabi simulation results for 1991–92 using on demand irrigation.

| Potential production million | Actual production million | Rainfall mm | Actual evapo-transpiration mm | Irrigation mm | Canal release $Mm^3$ | Tail loss $Mm^3$ |
|---|---|---|---|---|---|---|
| 0.143 | 0.110 (77%)** | 4 | 187 | 101 | 120.0 | 48.14 (40%)!! |

**% of potential production
!!% of canal release

of the kharif season. Table 4 summarizes the simulation results. It is seen that the actual production is 77% of the potential production. The actual production remains low due to limited canal release and negligible amount of rainfall during the season. The tail end loss is seen to be 40% of the canal release, which is significantly less than the figures for the kharif season (Table 3), but still rather high. This reflects the hydraulic design of the canal system due to which a relatively high water level (and flow) is required in order to divert water to the distribution channels. To compare the simulated crop production figures with the existing conditions in the command, a simulation has also been made for the rabi season without irrigation. It appeared that the water deficits without irrigation become more than 50% in almost all cases. This violates the basic assumption of the FAO relationship (Doorenbos & Kassam 1979) and consequently the yield calculations are not realistic. However, this shows, in accordance with known field practice, that no significant agricultural production is possible during the rabi season without irrigation, and further that even a relatively small amount of irrigation in the rabi season can provide the basis for a substantial crop production.

Figure 13 presents the effect of soil type and location of field along the main canal on the moisture content in the unsaturated zone. It is seen that heavy soils retain more moisture compared to light soils as expected. The moisture content, however, reduces significantly from upstream (DC 1) to downstream (DC 16) end of the main canal. This shows the nonuniform distribution of irrigation water along the length of the main canal, a common problem encountered in most of the irrigation commands with short supply.

Figure 14 presents the typical variation of moisture stress for different crops under similar soil and irrigation conditions. As expected, different crops undergo varying degree of moisture stress depending on their root depth and leaf area index development. However, it is seen that potato and mustard undergo higher moisture stress compared to wheat and gram.

In the light of above discussion, it is possible to enhance the actual production further by considering only the upstream command for cultivation,

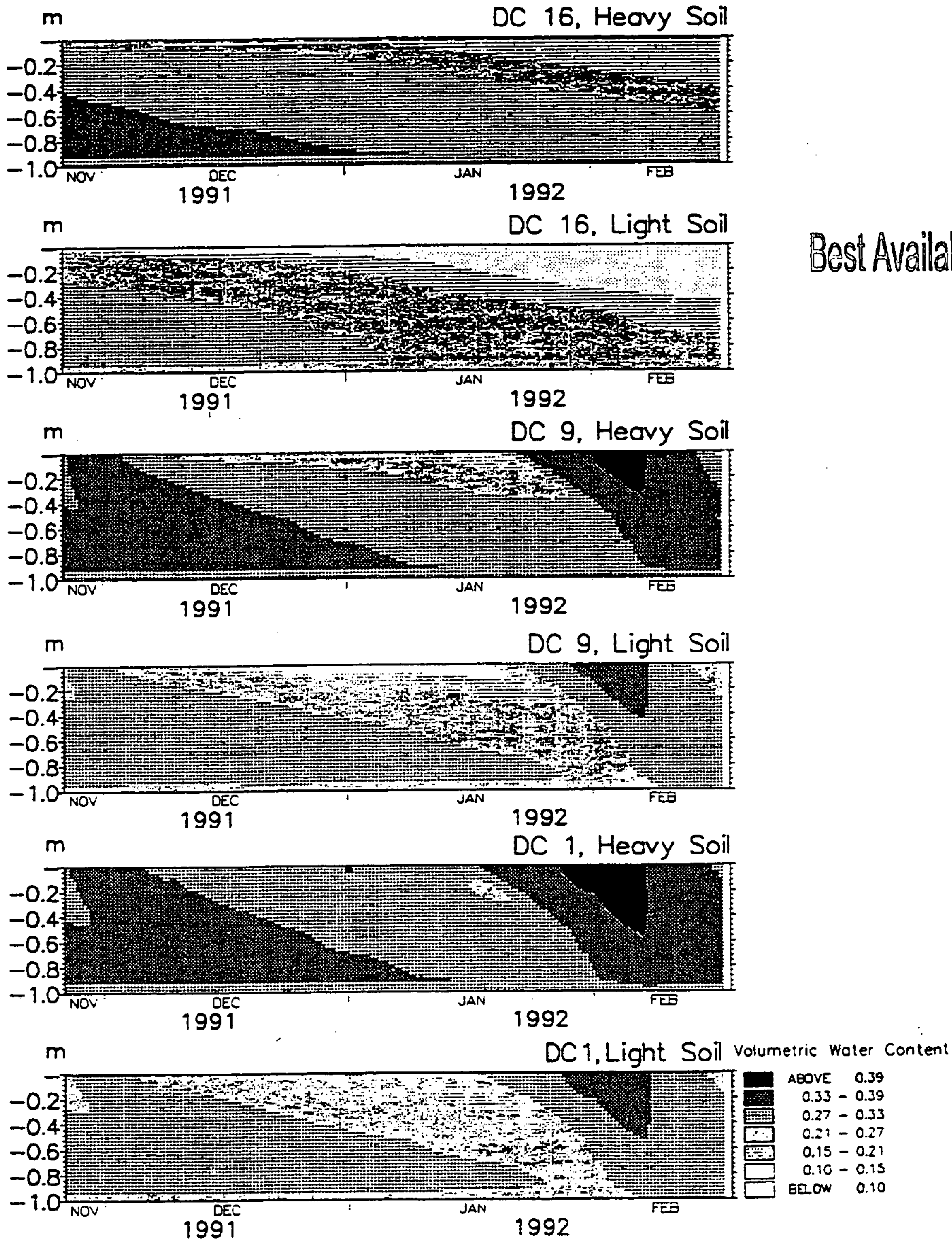




*Figure 13*. Typical variation of the unsaturated zone water content along the main canal during the rabi season. The locations of the distribution channels DC 1, DC 9 and DC 16 are shown in Fig. 5.

putting more area for cultivation under heavy soils and by selecting crops having higher moisture stress resistance. However, these are not attempted here. Nevertheless, the present modelling system provides the project management with a tool to study the different scenario and choose the most attractive

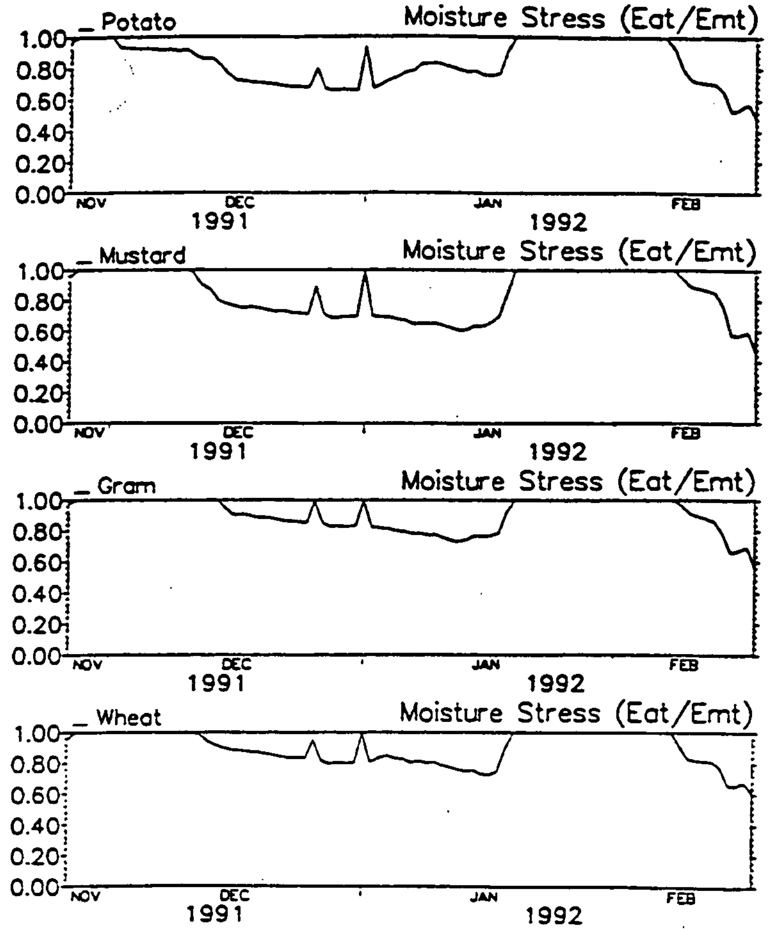


*Figure 14.* Typical variation of the moisture stress for different crops during the rabi season.

option. Further, it would prove to be more beneficial when planning for both kharif and rabi seasons taken together.

## Discussion and conclusions

The integrated hydraulic-hydrological modelling system presented in the present paper is a very comprehensive and versatile tool with many potential types of applications within the field of irrigation water management of command areas. The integrated modelling system differs from other systems presented in the litterature by providing detailed descriptions on both channel hydraulic, hydrological soil moisture and crop production aspects. The key elements of the integrated system, namely the MIKE 11 and MIKE SHE systems are well proven tools. The main limitation of the integrated modelling system is that the computational requirements, as a consequence of the general and advanced process descriptions, may be rather high for some types of applications. This is a practical problem today, but will gradually be compensated by faster computers.

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The case study, which illustrates one possible application of the integrated modelling system, has some important limitations, but shows nevertheless some interesting results both with respect to irrigation managment issues in the MRP and to more general issues.

In the case study, a macro-approach is adopted by lumping the branch canals and distributaries. Consequently, the field losses due to uneven distribution of surface water irrigation are not taken explicitly into account. In principle, this is feasible through a detailed discretization of the command as demonstrated by Lohani et al. (1993), who used the SHE to plot and field scales focussing on the variation of soil moisture from the head to the tail end of an irrigated field. However, in practice this would require exorbitant computer power and with the present generation of computers such a detailed study is not feasible for a large command area like MRP.

Another limitation is the assumption that the crops are transplanted or sown on the same day all over the command though under actual field conditions this is done over two to three weeks period. This implies that the model results from each of the 64 fields cannot be validated by a direct comparison with the local field conditions. Furthermore, for operational application data from more than one rainfall station in such large area is generally recommended. However, from a qualitative assessment the results appear to provide a reasonable representation of the field conditions in MRP.

A final important limitation of the case study is that the available data were not sufficient to allow a detailed calibration nor validation of the model. Hence, before such model is used operationally by irrigation managers more detailed data would be required, amongst others in terms of water level and discharge time series for several sites in the channel system. However, the overall order of magnitude of the simulated values (e.g. tail end loss) corresponds well with the perception of the MRP officials.

The results from the case study shows that presently a significant amount of water is wasted in the command area during the monsoon (kharif) season. It is demonstrated that a reduction of this waste could easily be made, and that the saved water could lead to a substantial crop production in the subsequent dry (rabi) season.

The results obtained from the case study further demonstrate the capability of the present modelling system to undertake the hydraulic and hydrological simulations of a large irrigation project simultaneously. The results also show how different modules interact with one another, providing useful information on various aspects of the irrigation canal command including canal losses, irrigation water utilization, moisture status in the unsaturated zone and crop growth. The modelling system provides the option of rotational or on demand irrigation. Though the rotational schedule is more popular in the developing countries, the on demand irrigation option can help in developing the guidelines for the former. The present modelling system, therefore, provides the irrigation managers with a versatile tool that can be used in planning and operation of large irrigation projects leading to better water use efficiency and improved crop production.

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# Parameterisation, calibration and validation of distributed hydrological models

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**Abstract**

This paper emphasizes the different requirements for calibration and validation of lumped and distributed models. On the basis of a theoretically founded modelling protocol, the different steps in distributed hydrological modelling are illustrated through a case study based on the MIKE SHE code and the 440 km$^2$ Karup catchment in Denmark. The importance of a rigorous and purposeful parameterisation is emphasized in order to get as few "free" parameters as possible for which assessments through calibration are required. Calibration and validation using a split-sample procedure were carried out for catchment discharge and piezometric heads at seven selected observation wells. The validated model was then used for two further validation tests. Firstly, model simulations were compared with observations from three additional discharge sites and four additional wells located within the catchment. This internal validation showed significantly poorer results compared to the calibration/validation sites. Secondly, the validated model based on a 500 m model grid was used to generate three additional models with 1000 m, 2000 m and 4000 m grids through interpolation of model parameters. The results from the multi-scale validation suggested that a maximum grid size of 1000 m should be used for simulations of discharge and ground-water heads, while the results deteriorated with coarser model grids.

## 1. Introduction

Hydrological models may be classified according to the description of the physical processes as conceptual and physically based, and according to the spatial description of catchment processes as lumped and distributed (Refsgaard (1996) and many others). In this respect, two typical model types are the lumped conceptual and the distributed physically based ones. Typical examples of lumped conceptual model codes are the Stanford Watershed Model (Crawford and Linsley, 1966) and the Sacramento (Burnash, 1995). The first outline of a distributed physically based model was made by Freeze and Harlan

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(1969). Since then, several codes have been developed such as the SHE (Abbott et al., 1986a, b; Bathurst and O'Connell, 1992; Refsgaard and Storm, 1995), the IHDM (Beven et al., 1987) and the Thales (Grayson et al., 1992a, b). A code such as TOPMODEL (Beven et al., 1995) may be characterized as conceptual distributed.

General methodologies related to model calibration, verification and validation have been subject to considerable discussion and dispute during the past decade, e.g. by Beven (1989), Bergström (1991), Tsang (1991), Konikow and Bredehoeft (1992), De Marsily et al. (1992) and Oreskes et al. (1994). However, as noted by Hassanizadeh and Carrera (1992) no consensus on methodology (or terminology) exists. Most of this scientific discussion has been of a principal nature and only a few authors, such as Klemes (1986), Anderson and Woessner (1992), IAHR (1994) and Refsgaard (1996), have attempted to outline general rigorous operational procedures.

Whereas much attention during the past three decades has been given to specific procedures for parameter assessment, calibration and, to a lesser extent, validation of lumped models (e.g. Fleming, 1975; WMO, 1975, 1986, 1992; Klemes, 1986; Sorooshian et al., 1993), very limited attention has so far been devoted to the far more complicated tasks in connection with distributed models, where problems related to validation of internal variables and multiple scales also have to be considered.

Distributed hydrological models are structured to enable the spatial variations in catchment characteristics to be represented by providing data for a network of grid points. Often model applications require several thousands of grid points, each of which is characterized by several parameters and variables. In this way distributed models differ fundamentally from lumped models, where a catchment is considered as one unit characterized by, typically, a few tens of parameters and variables. Thus the number of parameters and variables in a distributed model is, in principle, often two or three orders of magnitude higher than it would be for a lumped model of the same area. Obviously, this generates different requirements to lumped and distributed models with regard to parameterisation, calibration and validation procedures.

A critique expressed against distributed models by several authors concerns the many parameter values which can be modified during the calibration process. Beven (1989, 1996) considers models which are usually claimed to be distributed physically based as in fact being lumped conceptual models, just with many more parameters. Hence, according to Beven (1996) a key characteristic of the distributed model is that ''the problem of overparameterisation is consequently greater''.

Grayson et al. (1992a, b) indicating the enormous amount of information and theoretical potential of distributed physically based model codes such as THALES, emphasize the problems relating to code verification and model validation owing to difficulty in measuring/deriving parameters a priori and in measurement of catchment response in sufficient detail for testing. Grayson et al. (1995) emphasize ''the importance of evaluating distributed model behaviour rather than an integrated value such as runoff, when assessing the performance of distributed parameter models''.

The problems related to initialization, calibration and validation of distributed models are excellently summarized by Rosso (1994):

> ''In principle, spatially distributed models can accept experimental data at each grid element or calculation node. In practice, because of heterogeneity of parameter

values, differences between measurement scales and model grid scales, and experimental constraints, the specification of parameter values is very difficult. These constraints also apply to the validation of distributed model predictions by using measurements of internal system response. Conventional strategies for distributed model validation typically rely on the comparison of simulated model variables to observed data for specific points representing either external boundaries or intermediate locations on the model grid... Traditional validation based on comparing simulated with observed outflows at the basin outlet still remains the only attainable option in many practical cases. However, this method is poorly consistent with spatially distributed modelling..."

Refsgaard and Storm (1996) emphasize that a rigorous parameterisation procedure is crucial in order to avoid methodological problems in the subsequent phases of model calibration and validation. In parameterisation, the spatial patterns of the parameter values are defined so that a given parameter only reflects the significant and systematic variation described in the available field data, as exemplified by the practice of using representative parameter values for individual soil types, vegetation types or geological layers. Thus the parameterisation process effectively reduces the number of free parameter coefficients which need to be adjusted in the subsequent calibration procedure. The following points are important to consider in the parameterisation procedure (Refsgaard and Storm, 1996).

- The parameter classes (soil types, vegetation types, climatological zones, geological layers, etc.) should be selected so that it becomes easy, in an objective way, to associate parameter values. Thus the parameter values in the different classes should, to the highest possible degree, be assessable from available field data.
- It should explicitly be evaluated which parameters can be assessed from field data alone and which need some kind of calibration. For the parameters subject to calibration, physically acceptable intervals for the parameter values should be estimated.
- The number of real calibration parameters should be kept low, both from practical and methodological points of view. This can be done, for instance, by fixing a spatial pattern of a parameter but allowing its absolute value to be modified through calibration.

The aim of the present paper is to illustrate and discuss the problems associated with parameterisation, calibration and validation of distributed models. This is done by adapting a rigorous methodology and illustrating its use in a case study. The terminology and methodology used are defined in Section 2. The different steps and associated assumptions of parameterisation, calibration and validation of a distributed model are illustrated through a case study in Section 3. The case study does not focus on presenting good curve fitting, but on emphasizing fundamental points with regard to multi-criteria and multi-scale model validation. Finally, the generality of results are discussed in Section 4 with focus on parameterisation and calibration aspects and on the different validation requirements for lumped and distributed models.

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## 2. Terminology and methodology

The terminology applied in the present paper follows that described in Refsgaard (1996). The modelling methodology is illustrated by the modelling protocol presented in Fig. 1, which is adapted from Anderson and Woessner (1992). Terminology-wise, a distinction is made between a model and a model code. A model is defined as a particular

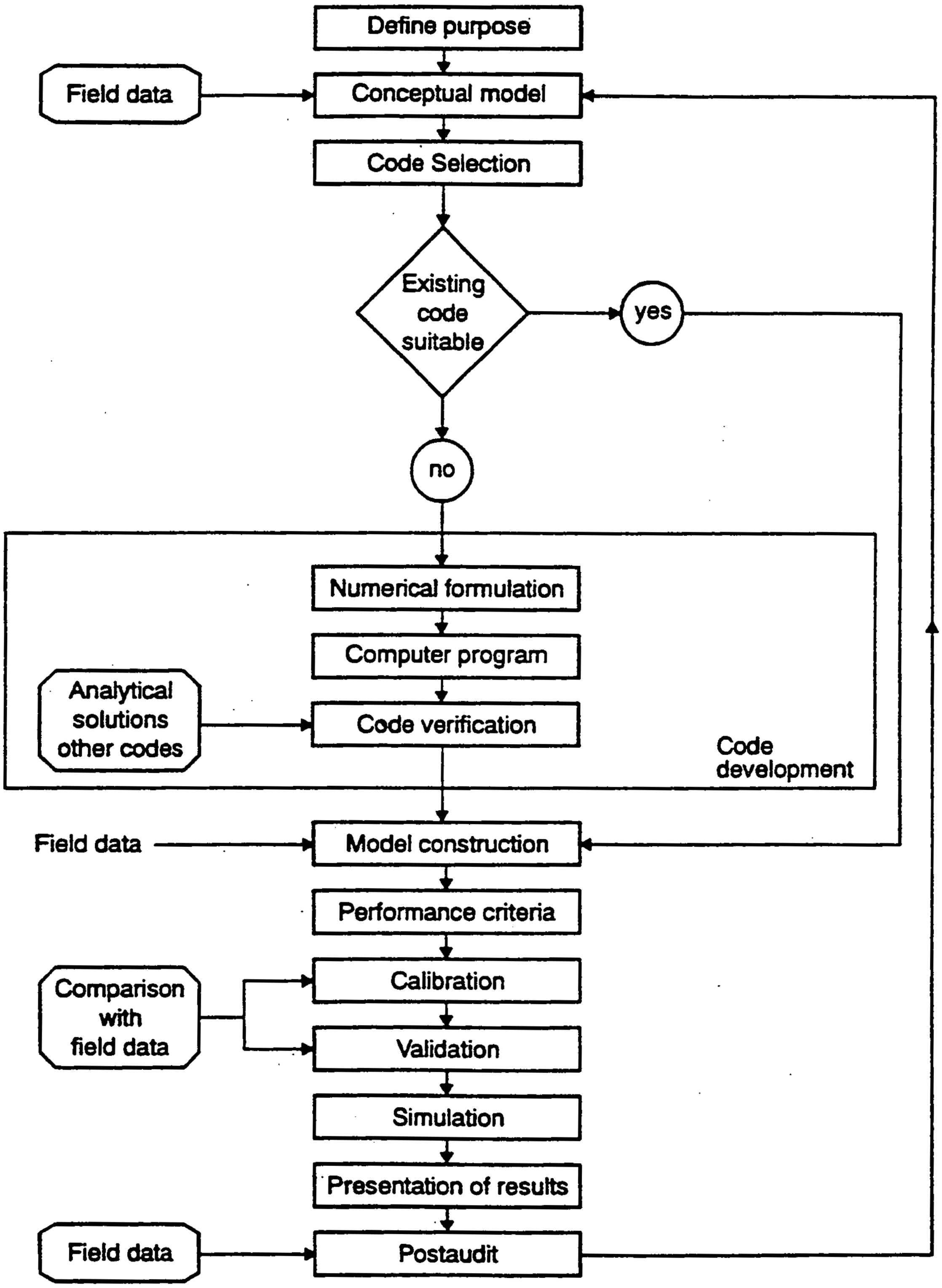


Fig. 1. The different steps in hydrological modelling—a modelling protocol.

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hydrological model established for a particular catchment. A model code, however, is defined as a generalized software package, which, without program changes, can be used to establish a model with the same basic types of equations (but allowing different parameter values) for different catchments.

Model validation is here defined as the process of demonstrating that a given site-specific model is capable of making sufficiently accurate predictions. This implies the application of the calibrated model without changing the parameter values that were set during the calibration, when simulating the response for a period other than the calibration period. The model is said to be validated if its accuracy and predictive capability in the validation period have been proven to lie within acceptable limits.

## 3. Case study: Karup catchment, Denmark

The purpose of the case study is to illustrate the different steps and the inherent assumptions in distributed modelling in general, and to focus on special requirements of validation of internal variables and, in particular, on the effects of using different model discretizations.

### *3.1. The Karup catchment*

The 440 $km^2$ Karup catchment (Fig. 2) is located in a typical outwash plain in the western part of Denmark. From a geological point of view, the area is relatively homogeneous, consisting of highly permeable sand and gravel with occasional lenses of moraine clay.

The land use in the catchment consists of agriculture (67%), forest (18%), heath (10%), wetland (4%) and urban (1%). The catchment has a gentle sloping topography and is drained by the Karup River and about 20 tributaries. Owing to the highly permeable soils, all water outside the wetland areas infiltrates, and the discharge regime is dominated by baseflow. The catchment area defined by the topographical divide is slightly larger than the area determined from the ground-water divide. However, as no overland flow occurs and no streams exist in the area outside the ground-water divides (Stendal, 1978), the model has, in line with previous studies, been confined to cover the 440 $km^2$ ground-water catchment.

The depth of the unsaturated zone varies from 25 m at the eastern ground-water divide to less than 1 m in the wetland areas along the main river. The aquifer is mainly unconfined and of glacial deposits. The thickness of the aquifer varies from 10 m in the western and central parts to more than 90 m at the top of the outwash cone to the east.

The Karup catchment has been subject to comprehensive hydrological studies. Firstly, it was one of the Danish representative basins under IHD, and a comprehensive data collection programme was conducted in 1965–1977 (Stendal, 1978). Secondly, the IHD data formed the basis for a comprehensive hydrological modelling investigation in the beginning of the 1980s, aimed at assessing the impacts on streamflow of ground-water abstraction for irrigation (Miljøstyrelsen, 1983). Finally, a major research project focusing on nitrate pollution from agriculture was conducted in the area during the second half of the

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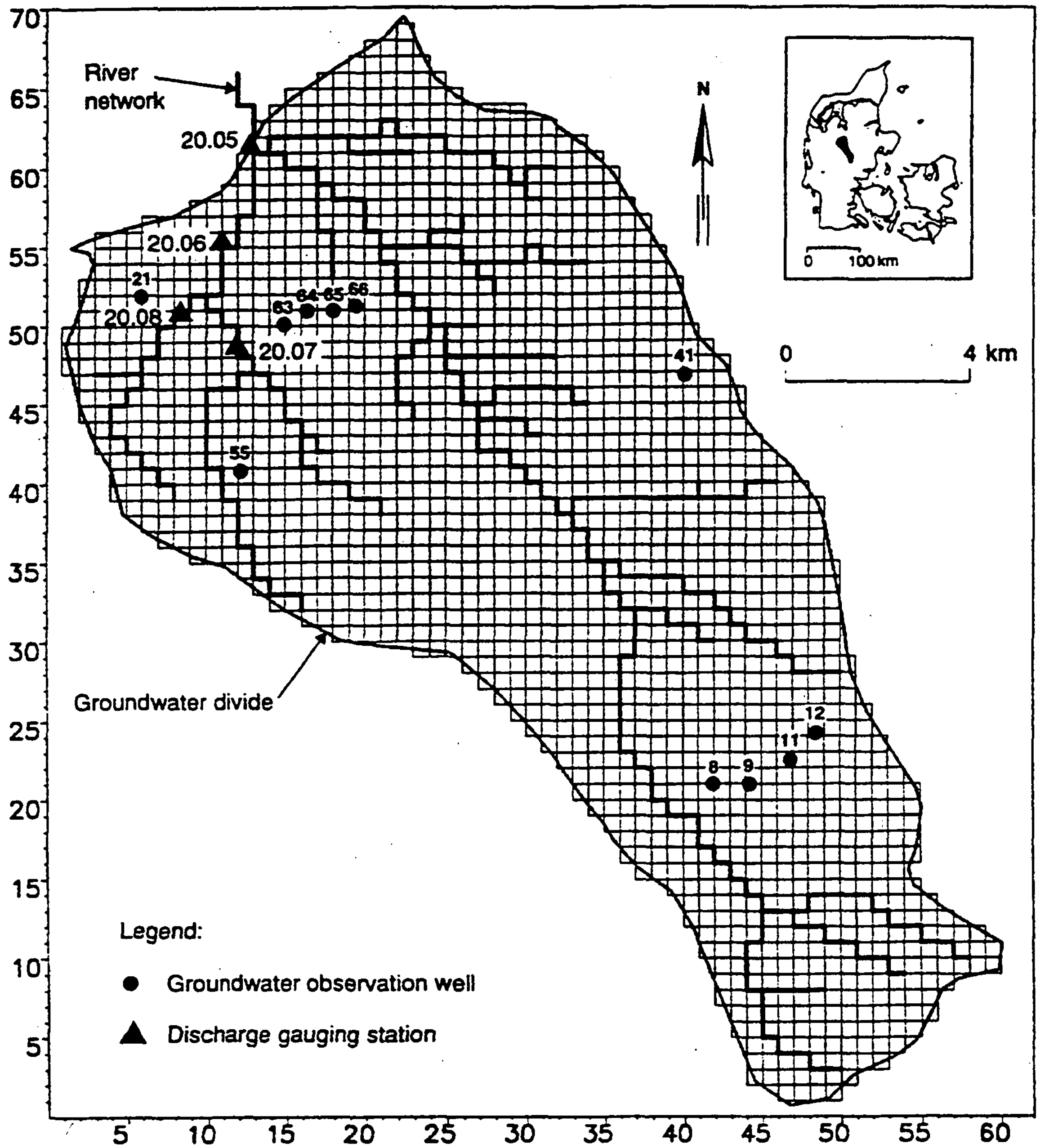


Fig. 2. The Karup catchment with the river network in a 500 m model grid together with the locations of the discharge gauging stations and groundwater observation wells referred to in the text.

1980s (Miljøstyrelsen, 1991). Styczen and Storm (1993), as part of the nitrogen research project, developed a model for the Karup catchment capable of simulating hydrology, nitrogen leaching and nitrogen movements at catchment scales. The present case study is based on data processed by Styczen and Storm (1993), but the hydrological model in the present case study is simpler (2D versus 3D ground-water; four vegetation/crop classes instead of 18, etc.) and the calibration and validation is mostly carried out independently of Styczen and Storm (1993).

### 3.2. Modelling

The 11 steps in the modelling protocol outlined in Fig. 1 have been carried out as follows.

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*3.2.1. Step 1: definition of purpose*

The overall objectives of the case study are to illustrate the parameterisation, calibration and validation of a distributed model and to study the validation requirements with respect to simulation of internal variables and to changing spatial discretization. In this context, the purpose of the model is to simulate the overall hydrological regime in the Karup catchment, especially the dynamics of discharges and ground-water tables.

*3.2.2. Step 2: establishment of a conceptual model*

It is emphasized that it is not an objective to establish the most detailed and best possible hydrological model for the area by making use of all available data, but rather to carry out a methodologically rigorous modelling study illustrating the consequences of a typical distributed modelling approach for simulation of discharges and ground-water tables. Therefore, some simplified assumptions are made with regard to the conceptual model, as compared to previous model studies with other objectives, such as Styczen and Storm (1993). A conceptual model comprises the user's perception of the key hydrological processes in the catchment and the corresponding simplifications which are assumed to be acceptable in the mathematical model in order to achieve the purpose of the modelling. In the present case, the conceptual model comprised the following elements.

*3.2.2.1. Hydrogeology.* The basis for the assessment of hydrogeological parameters is a detailed geological description based on 20 cross-sectional profiles (15 in the SW–NE direction and five in the NW–SE direction, interpreted from 140 well-logs) covering the entire catchment (Hansen and Gravesen, 1990). The aquifer is assumed to consist of one main aquifer material characterized by the same hydraulic parameters throughout the catchment and five lenses with distinctly different hydraulic parameters. The aquifer is assumed to be unconfined. In accordance with a previous model study (Miljøstyrelsen, 1983), it was assumed that a two-dimensional ground-water model would be sufficient for simulation of discharges and ground-water levels.

*3.2.2.2. Soil and unsaturated zone.* Soil maps supported by profile descriptions and soil analyses (texture, density, retention curves) exist for the entire catchment. Two soil profiles were used in the model. For the main part of the area, the soil profile "general" (Fig. 3) was used, comprising loamy sand to a depth of 100 cm and fine sand below. For heath areas, the soil profile "heath" was used, comprising fine sand for the upper 55 cm and coarse sand below. The depth of the unsaturated zone is assessed simply by comparing the topography with the location of the ground-water table. The water is assumed to flow vertically in the unsaturated zone, and owing to the sandy soil, macropore/bypass flow is assumed to be negligible. In the main parts of the area, the water movements are downwards as ground-water recharge, while upward capillary flux occurs in wetland areas during dry summer periods.

*3.2.2.3. Vegetation/crops.* Four vegetation/cropping classes were assumed: agricultural, forest, heath and wetland grass. The spatial distribution of the four classes, shown in Fig. 3, is taken from Styczen and Storm (1993), with the modification that all the 15 agricultural crop classes used for describing fertilization practises and nitrogen leaching were lumped

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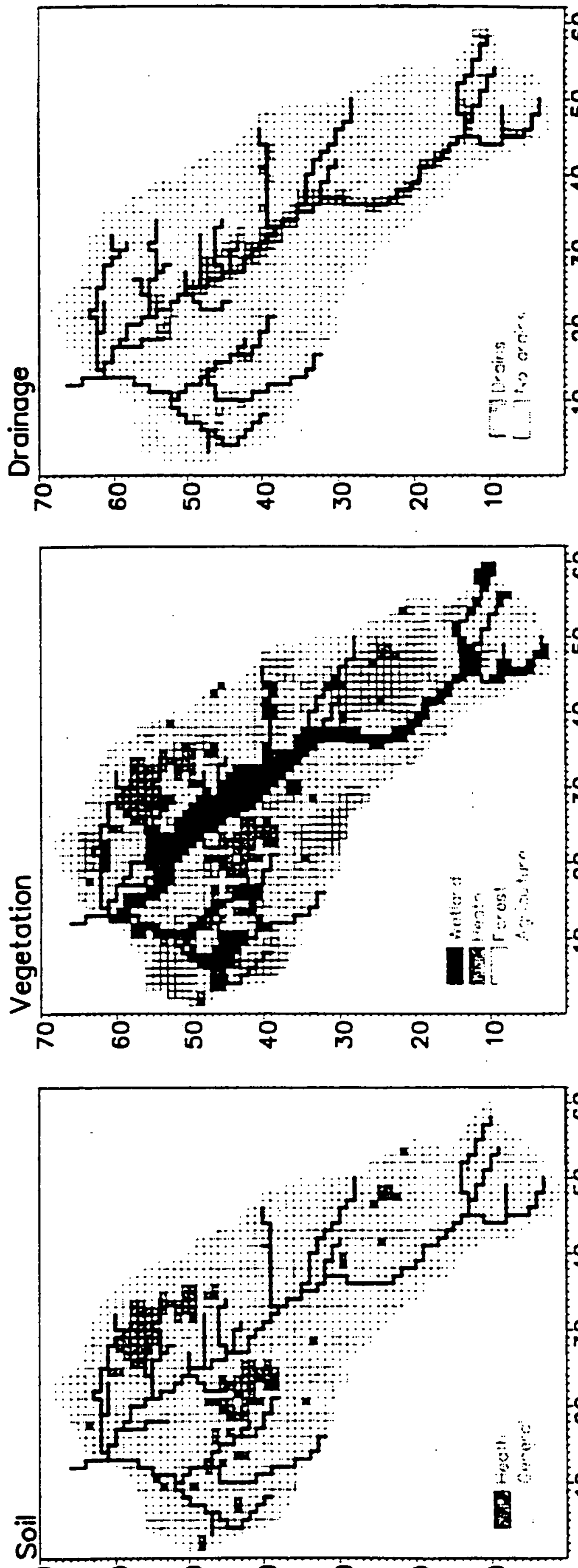


Fig. 3. Soil, vegetation and drainage maps used in the model.



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into one, which is assumed to be sufficient for estimation of evapotranspiration and recharge.

*3.2.2.4. Surface water drainage system.* The main river and the tributaries which could be accommodated within the spatial model discretization are included in the model. In addition, significant parts of the wetland areas near the main river are known to be drained by ditches and tile drain pipes. The extent of the wetland areas shown in Fig. 3 has been assessed from topographic maps.

*3.2.2.5. Stream–aquifer interaction.* A thin, low, permeable layer is assumed to exist between the river and the main aquifer. The leakage coefficient characterizing this layer will be subject to calibration. From earlier studies (Storm and Refsgaard, 1980; Miljøstyrelsen, 1983), it was known that a prerequisite for adequately simulating the dynamics of the stream–aquifer interaction is to have a sufficiently fine spatial model resolution. The critical aspect, in this regard, is to be able to resolve the topographical variation of the river valley, which is typically 500–2500 m wide and which at the downstream parts of the catchment, is 10–15 m deep as compared to the surrounding terrain.

*3.2.2.6. Climate.* Daily values of precipitation were available from nine stations. Furthermore, sunshine hours and average temperature were available from stations four and one, respectively. The variability between stations is relatively small (standard deviation of annual precipitation about 3% of mean). Hence, for the present purpose it was assumed that areally average values would be sufficient, so daily means were calculated for precipitation, potential evapotranspiration and temperature.

### *3.2.3. Step 3: selection of model code*

The MIKE SHE code (Refsgaard and Storm, 1995) was selected for the case study. MIKE SHE is a distributed physically based code with an integrated description of the entire land phase of the hydrological cycle. It comprises components for overland flow (two-dimensional, kinematic wave), river flow (one-dimensional, diffusive wave), unsaturated flow (one-dimensional, Richards' equation), interception (Rutter model), evapotranspiration (Kristensen and Jensen model), snowmelt (degree–day approach), saturated flow (two- or three-dimensional Boussinesq). MIKE SHE is able to address all the requirements defined by the conceptual model.

### *3.2.4. Step 4: code verification*

As MIKE SHE is a well proven code with several verification tests as well as many large-scale, engineering applications including prior tests on the present area and on similar cases, no additional code verification was required in this case.

### *3.2.5. Step 5: model construction and parameterisation*

Model construction involves designing the model with regard to the spatial discretisation of the catchment, setting boundary and initial conditions and making a preliminary

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selection of parameter values from the field data. An important aspect in this respect is the parameterisation.

*3.2.5.7. Discretization.* The Karup catchment was divided into grid squares of 500 m × 500 m. The vertical division varied between 5 cm in the topnodes to 40 cm in the lower part of the soil profile in the unsaturated zone.

*3.2.5.8. Topography.* The topographical contour data were digitized from 1:50 000 maps and interpolated to obtain average elevations for each grid square.

*3.2.5.9. River and drainage system.* The stream system was digitized and bank elevations assigned to specific points along the river course. In addition, cross-sections were assessed at specific locations in the stream system. A Manning number (river bed resistance) of 20 $m^{1/3}$ $s^{-1}$ was used throughout the area. As the hydrograph is dominated by slowly varying baseflow, the river routing itself has no significance for the shape of the hydrograph. Hence the Manning number was not subject to calibration. The Manning number, however, does have some influence on the river water level and hence on the stream–aquifer interaction. As the dynamics of this interaction are also significantly affected by the leakage coefficient of the river-bed material, this last coefficient was chosen for calibration. For the wetland areas with artificial drainage in terms of ditches and pipes at a scale much finer than the 500 m grid, drains were introduced into the model at a depth of 1.0 m below the ground surface. The runoff from these ''model drains'' starts when the groundwater levels are above the drain depths and is proportional to this height difference. The drainage parameter, corresponding to a time constant in a linear reservoir (i.e. outflow linearly proportional to storage), is subject to calibration.

*3.2.5.10. Aquifer system.* The three-dimensional geological model comprising the main aquifer and lenses was digitized. The hydraulic parameters for the lenses (conductivities and storage coefficients) were assessed beforehand from hydrogeological data and were not subject to calibration (approach like that of Styczen and Storm (1993)). Furthermore, owing to the coupled unsaturated–saturated zone description in MIKE SHE, where the two zones overlap each other, the specific yield of the aquifer is, in reality, a passive parameter, which is determined by the soil moisture retention curve of the corresponding layer of the unsaturated zone. Thus, the only groundwater parameter which was calibrated was the hydraulic conductivity of the main aquifer material. Information on this sensitive parameter was also available from field pumping tests and previous modelling studies; however with all the simplifications made in the geological model (2D instead of 3D, constant hydraulic conductivity in space, etc.) it was necessary to fit this parameter through calibration. As only two-dimensional groundwater modelling was required, the data on the main aquifer material and on the different lenses were used to derive depth-averaged two-dimensional hydraulic parameter values through integration over depth. The model boundaries were assumed to be impermeable. As the boundaries coincide with the natural groundwater divides, such a ''no-flow'' boundary condition can be justified. The groundwater abstraction, in accordance with Miljøstyrelsen (1983), has been assumed to be negligible.

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*3.2.5.11. Soil.* The soil water retention curves for the four soil types were taken directly from measurements, while no reliable measurements for unsaturated hydraulic conductivities existed. Therefore, a theoretical formula (Brooks and Corey, 1964) describing the conductivity as a function of soil moisture content was used:

$$K(\theta)=K_{\mathrm{sat}}\left(\frac{\theta-\theta_{\mathrm{res}}}{\theta_{\mathrm{sat}}-\theta_{\mathrm{res}}}\right)^{n}$$

where: $K(\theta)$ is hydraulic conductivity; $\theta$ is actual soil moisture content; $K_{\mathrm{sat}}$ is hydraulic conductivity at saturation; $\theta_{\mathrm{sat}}$ is moisture content at saturation, derived from retention curve; $\theta_{\mathrm{rel}}$ is residual moisture content, estimated to 1 vol.%; and $n$ is exponent.

For each of the four soil types, two parameters were subject to calibration, namely $K_{\mathrm{sat}}$ and $n$. The unsaturated hydraulic conductivities in the root zone have significant influence on the soil moisture contents and hence the actual evapotranspiration. As the vegetation and evapotranspiration parameters were not calibrated (see below), the aim of the calibration of the soil hydraulic parameters was to ensure a good simulation of the overall water balance for the catchment. This implies that $K_{\mathrm{sat}}$ and $n$ become fitted parameters which, owing to no independent checks of the vegetation parameters, incorporate possible biases from these and, as such, lose some of their direct physical interpretation capabilities.

*3.2.5.12. Vegetation.* For each of the four vegetation/crop classes time series of leaf area index and root depth were defined. The leaf area index curves were simulated by the DAISY model as described by Styczen and Storm (1993). The root depth time series were assessed from literature. Furthermore, the values of the empirical evapotranspiration and inception storage parameters were selected as being identical to values successfully used in many other modelling studies in Denmark and abroad. Altogether, no parameter values were subject to calibration.

*3.2.5.13. Initial conditions.* As explored for the first time by Stephenson and Freeze (1974), initial conditions are very important in this kind of modelling. In the Karup catchment, the large storage possibilities in the deep unsaturated zone and in the unconfined aquifer imply that the initial conditions influence the simulation results for several years. Thus, the groundwater table in addition to the annual fluctuations clearly shows long-term variations with phases in the order of a decade as responses to sequences of dry or wet years. The following approach was used for assessment of initial conditions valid for 1st January 1969.

1. The available data from observation wells were not sufficient for interpolation of initial conditions, which, therefore, had to be assessed in an iterative procedure using the model. According to available observation wells, the groundwater tables appeared to be approximately at the same level in December 1978 as in January 1969. Therefore, the 1969 initial conditions were derived from the 1978 model simulated values as follows: (1) a model run for the period 1969–1978 was made with guessed initial conditions for 1969; (2) simulated groundwater levels for December 1978 were extracted and used as initial conditions in a second model run; (3) if the simulated 1978 values in the two first runs are not identical, a third run is made on the basis of initial condition

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extracted from model results of the second run; etc. As the groundwater tables depend on the various model parameters and as consistency between model parameters and initial conditions is important, this procedure was repeated though the calibration process.

2. The water content in the unsaturated zone was assumed to correspond to field capacity ($pF = 2$).
3. The two years 1969 and 1970 were used as a "warm up period". Hence, model results for this period were not used in the calibration process.

### 3.2.6. Step 6: performance criteria

As similar modelling had been carried out for the same catchment earlier (Miljøstyrelsen, 1983; Styczen and Storm, 1993), definition of performance criteria were simpler than in many other cases, where specification of acceptable levels of accuracy prior to the first modelling runs was far from simple, but had to be related to the data availability and the specific purpose of the particular study.

In the present case, the performance criteria were not defined rigorously as numerical figures, but rather as the same level of accuracy as achieved by Styczen and Storm (1993). More specifically, the performance criteria were related to the following variables.

1. Discharge simulation at station 20.05 Hagebro (the outlet of the catchment) with a graphical assessment of observed and simulated hydrographs supported by the following two numerical measures:
    - average discharges of observed and simulated records, $OBS_{ave}$ and $SIM_{ave}$, and
    - model efficiency, R2, calculated on a daily basis (Nash and Sutcliffe, 1970).
2. Groundwater level simulations at observation wells 21, 44 and 55 located at the downstream part of the catchment and also used by Styczen and Storm (1993) plus observation wells 8, 9, 11, 12 representing a cross section at the upstream part of the catchment.

### 3.2.7. Step 7: model calibration

The following parameter values were subject to adjustments through the calibration phase.

1. The horizontal conductivity of the main aquifer material. A value of $3.5 \times 10^{-4}$ m s$^{-1}$ was obtained. This corresponds well with pumping test data: Miljøstyrelsen (1983) reports data from 14 pumping tests with hydraulic conductivities ranging from 0.5 to $5.6 \times 10^{-4}$ m s$^{-1}$.

Table 1
Soil hydraulic parameters assessed through calibration

| Soil profile | Soil depth | $K_{sat}$ (m s$^{-1}$) | $n$ |
|---|---|---|---|
| General | 0–100 cm | $1.0 \times 10^{-5}$ | 18.8 |
| | below 100 cm | $3.5 \times 10^{-5}$ | 10 |
| Heath | 0–55 cm | $3.5 \times 10^{-6}$ | 10 |
| | below 55 cm | $2.0 \times 10^{-4}$ | 6 |

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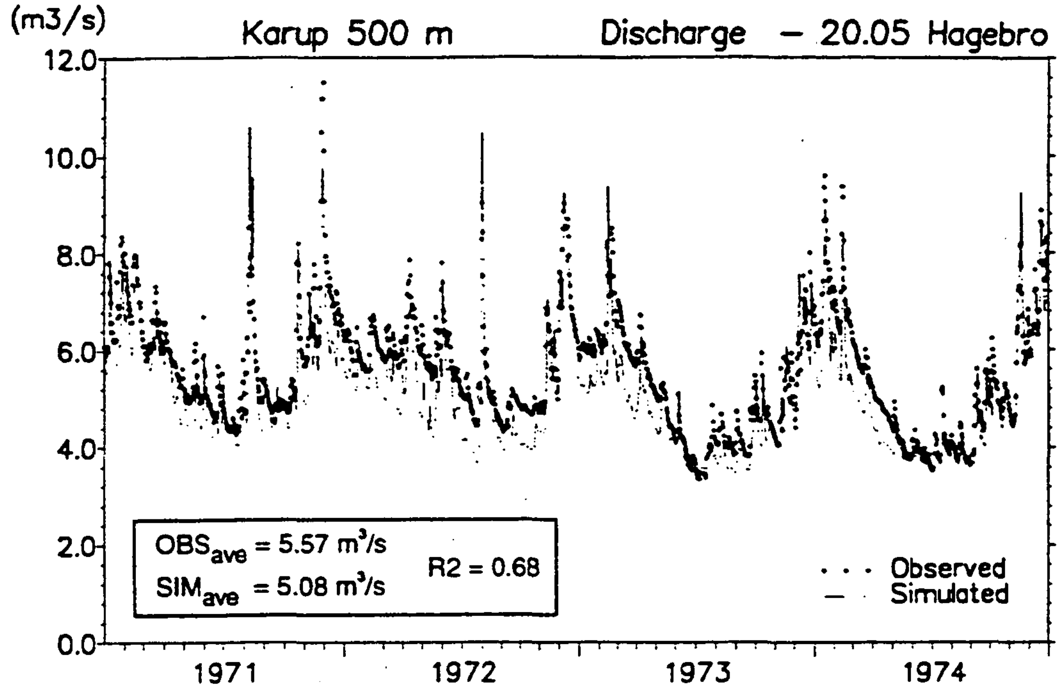


Fig. 4. Simulated and observed discharge for the entire catchment for the calibration period together with figur for average observed and simulated flows, $OBS_{ave}$ and $SIM_{ave}$, and model efficiency on a daily basis, R2.

2. The leakage coefficient of the river-bed material. A value of $3 \times 10^{-7}$ s$^{-1}$ was obtaine
3. A drainage coefficient for the wetland areas. A value of 0.03 per day corresponding to time constant of 33 days was obtained.
4. The eight soil hydraulic parameters shown in Table 1.

The model calibration was carried out on the basis of data for the period 1971–1974. maximum time-step of 6 h was used in the transient simulation. The time step wa however, automatically reduced by MIKE SHE in situations, where water balance erro above specified accuracy limits occurred in some of the model components. For exampl time steps were typically reduced to a few minutes in the unsaturated zone in connectic with heavy rainfall. Calibration results are shown in Figs 4 and 5 for discharge and groun water tables, respectively, and key water balance figures are shown in Table 2. Th dynamics and the mean levels of observed and simulated values are, in general, in reaso ably good agreement. An exception to this is the simulated baseflow which, for the fir half of the calibration period, is significantly below the observed flows resulting in a 9 underestimation of the average flow. This may originate from different causes, such uncertain estimates of initial conditions for ground-water levels. Altogether, however, th calibration results are of the same accuracy as the results in Styczen and Storm (1993), ar are, as such, considered to be acceptable.

*3.2.8. Step 8: model validation—split-sample*

A traditional split-sample validation test was conducted against data from the san discharge station and ground-water observation wells as used for calibration. The peric 1975–1978 was used for validation. The validation results are shown in Figs 6 and 7, fro which it appears that the validation results both with respect to hydrograph shapes, wat

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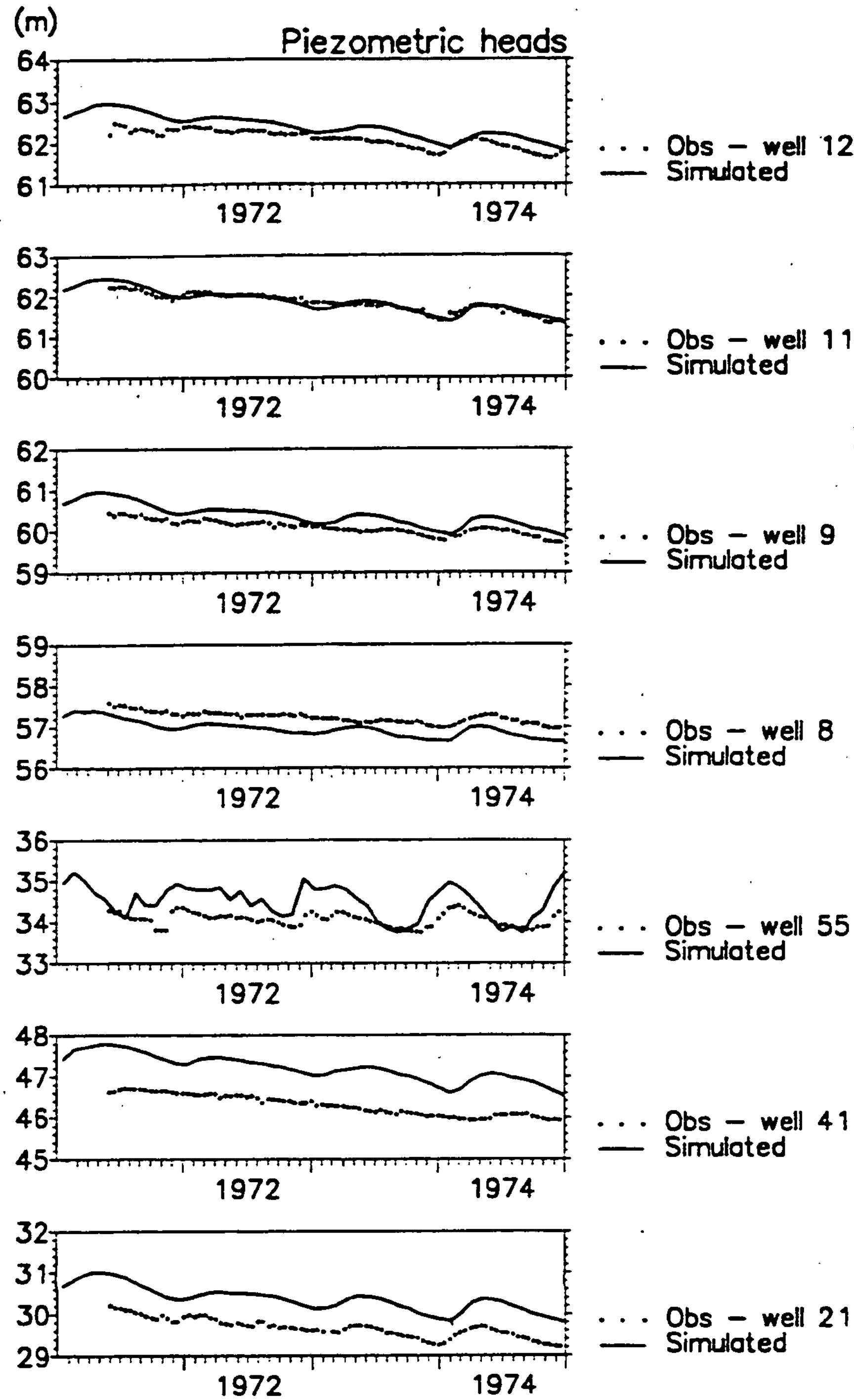




Fig. 5. Simulated and observed piezometric heads at seven well-sites for the calibration period. The locations of the wells are shown in Fig. 2.

Table 2
Key water balance figures (in mm per year) for the Karup catchment from the calibration and validation periods

| | Calibration 1971–1974 | Validation 1975–1978 |
|---|---|---|
| Precipitation | 809 | 796 |
| Potential evapotranspiration | 630 | 621 |
| Observed discharge | 399 | 366 |
| Simulated discharge | 364 | 373 |

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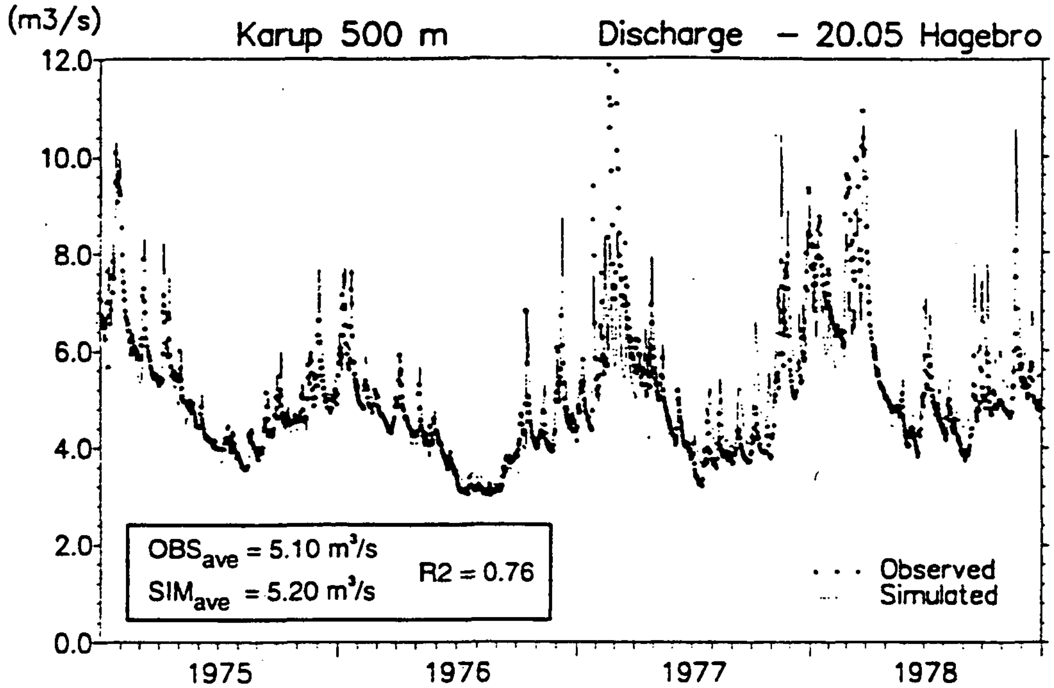


Fig. 6. Simulated and observed discharge for the entire catchment for the validation period together with figur for average observed and simulated flows, $OBS_{ave}$ and $SIM_{ave}$, and model efficiency on a daily basis, R2.

balance and model efficiency (R2 criteria) are of the same level of accuracy as tl calibration results.

*3.2.9. Steps 9–11: simulation, reporting, postaudit*

The outcome of the simulation step in the modelling protocol (Refsgaard, 1996) is tl modelling studies carried out with respect to multi-site and multi-scale validation. The: results are reported below.

*3.3. Model validation: multi-site*

The adopted validation test scheme, based on one discharge station and seven groun water observation wells corresponds, by and large, to the validation approach for lumpe models, except that a lumped model would generally not be appropriate for simulation ground-water tables. On the basis of the successful validation test it can be concluded th the model can be considered valid for simulation of the rainfall–runoff relationship for tl whole catchment and that the model, in addition to this, is valid for simulation of groun water tables at the seven observation wells. The model can be expected to be valid fi future conditions, provided that no significant non-stationarities, such as climate change changes in land use or water abstractions, occur.

Quite often, modellers, after having passed such validation test, would claim that tl distributed model is, now, valid also for simulation of internal flows and ground-wat tables internally in the catchment. For instance, such expectations were made t Refsgaard et al. (1992) and Jain et al. (1992) on catchments in India, where no da were available for internal validation.

In order to test the model's capability to simulate internal conditions, a multi-si validation test was carried out. This new test scheme comprised of comparisons

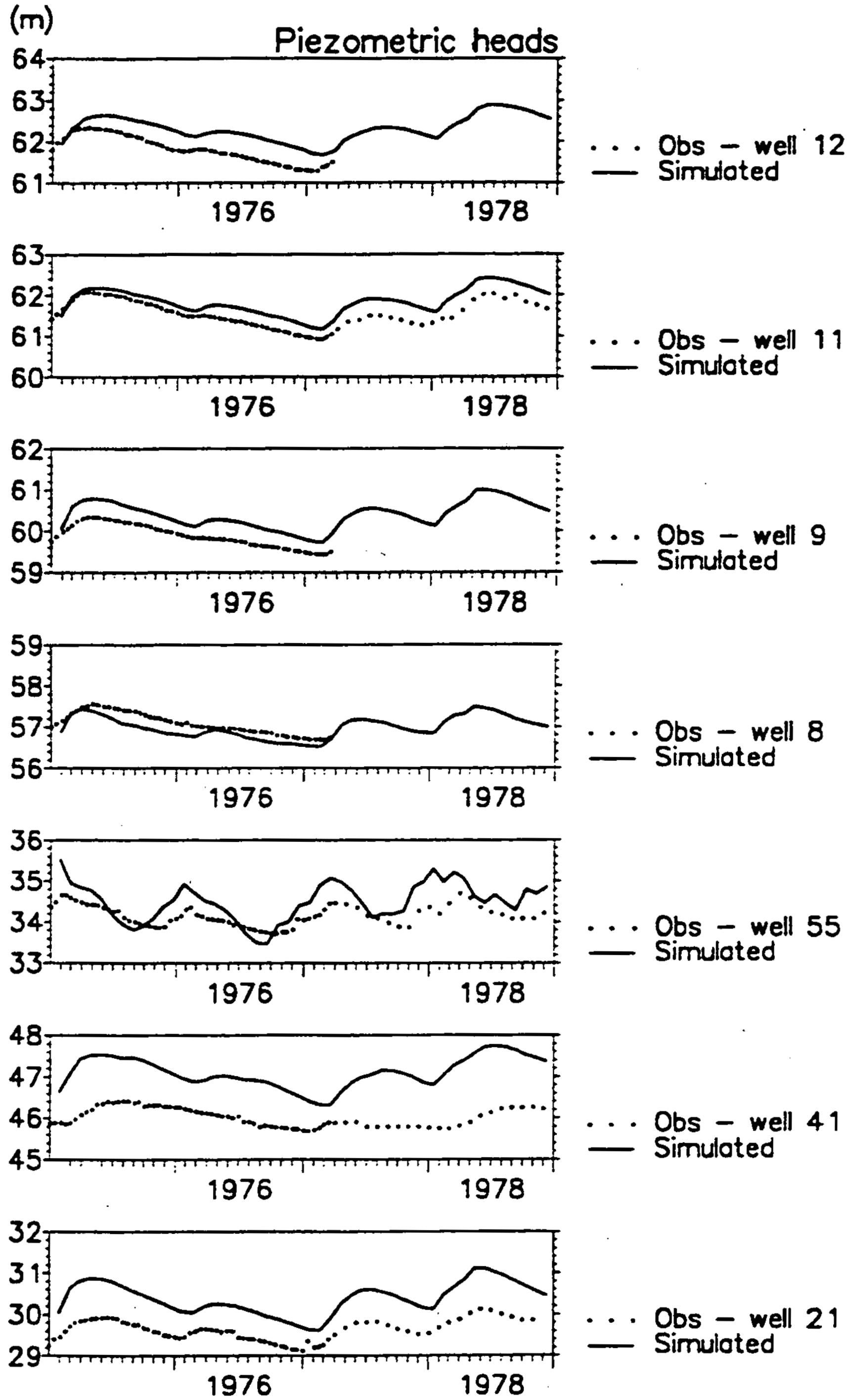


Fig. 7. Simulated and observed piezometric heads at seven well-sites for the validation period. The locations of the wells are shown in Fig. 2.

simulated and observed data for the following stations, for which data were not used at all during the calibration process:

1. discharge values at the three stations 20.06 Haderup (98 km$^2$), 20.07 Stavlund (50 km$^2$) and 20.08 Feldborg (17 km$^2$) (Fig. 2);
2. ground-water tables at observation wells 63, 64, 65 and 66, located in the area between the main river and the tributary with the three discharge stations 20.06, 20.06 and 20.07 (Fig. 2).

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Results from the first 28 months of the validation period, where data are available for all the above stations, are seen in Figs 8 and 9 for discharge and ground-water tables, respectively. As can be seen both from the hydrographs and the water balance and model-efficiency figures in Fig. 8, the simulation results are significantly less accurate than for the calibrated stations. The discharges at the three tributary stations are significantly more poorly simulated than for the calibrated station 20.05 in two respects. Firstly, there is a clear undersimulation of the baseflow level and the total runoff for the three tributary stations. Secondly, the simulation shows a somewhat more flashy response than the observed hydrographs. The primary reason for the differences in baseflow levels is that the internal ground-water divide between the main river and the main tributary is not simulated correctly, with the result that the three tributary stations, according to the model, are draining smaller areas than they do in reality. This may be explained by apparently inaccurately simulated ground-water levels in these areas, probably owing to the simplifications made in the geological model (2D, constant hydraulic conductivity of main aquifer, etc.). In addition to this, inaccurate boundary conditions may, especially for the smallest catchment located next to the overall catchment divide (station 20.08), play some role as well. The too-flashy model response is due to simulated drainage flow from wetland areas, and as such an indication of an incorrect time constant in the parameter value for drainage in this area, which again is an indication that the spatially constant value fitted through calibration against the entire catchment discharge (20.05) may not be representative for all subcatchments. The simulated ground-water tables (Fig. 9) show correct dynamics, but have problems with the levels supporting the above indication of a not very accurate simulation of the internal ground-water divide between the main river and the tributary.

### *3.4. Model validation: multi-scale*

In addition to simulation of internal variables, a distributed model is often used with different discretizations. Thus often a coarse grid is used in initial calibrations and subsequently the grid is refined for final calibrations. However, owing to scaling problems such an approach is often problematic. In the present case, sensitivity tests were made using the model (calibrated and validated on a 500 m grid) on coarser grids: 1000, 2000 and 4000 m.

The MIKE SHE code comprises routines for automatic generation of models on different grids. Thus, all the basic data with regard to geology, topography, vegetation types, etc. for the four models were identical. The only modifications made manually when generating the three coarser grid models were:

1. Some of the tributaries had to be deleted, because MIKE SHE only allows rivers between grids. Thus especially for the 4000 m grid many tributaries "disappeared" The topography and river network of the four models are shown in Fig. 10.
2. The initial conditions for the ground-water levels had to be estimated for each of the models through the procedure described in Section 3.2.5.13 above.

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The results of the four model-simulations of discharge for the entire catchment are shown in Fig. 11. It appears that the 500 m and the 1000 m models only differ marginally,

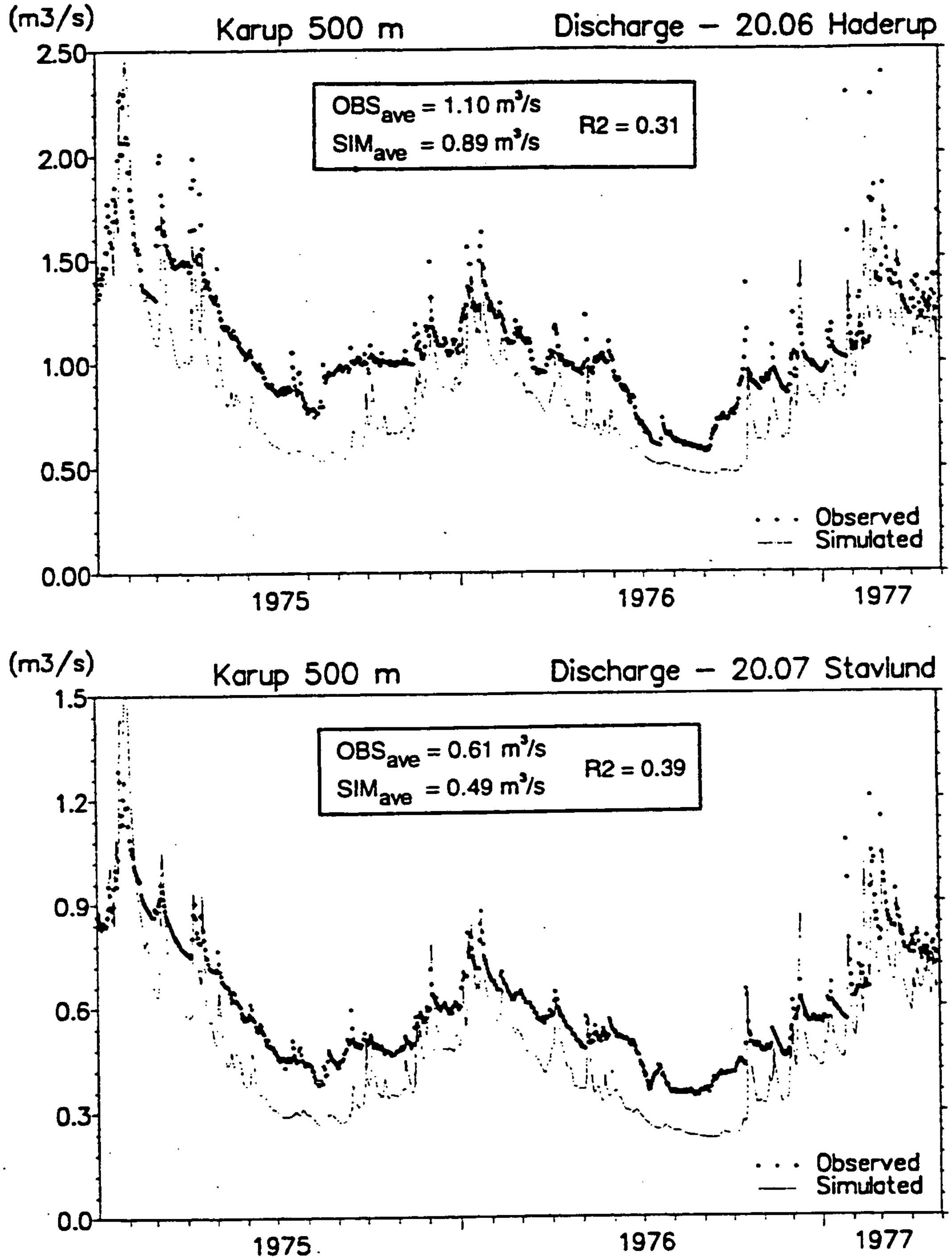


Fig. 8. Simulated and observed discharges, average flows, $SIM_{ave}$ and $OBS_{ave}$, and model efficiencies, R2, from the validation period for three internal discharge sites 20.06 (98 km$^2$), 20.07 (50 km$^2$) and 20.08 (17 km$^2$), which have not been subject to calibration. The locations of the discharge stations are shown in Fig. 2.

whereas the 2000 m and in particular the 4000 m model show a significantly less accurate runoff simulation as compared to the 500 m model. It is noticed that the runoff volumes, expectedly, are not varying much, but the runoff response, especially during dry summer periods such as 1975, is fundamentally different. The reason for this is believed to be the fact that the 2000 m and the 4000 m models are not able to give sufficiently fine resolution of the river valley (Fig. 10) and hence the stream–aquifer interaction cannot be correctly simulated.

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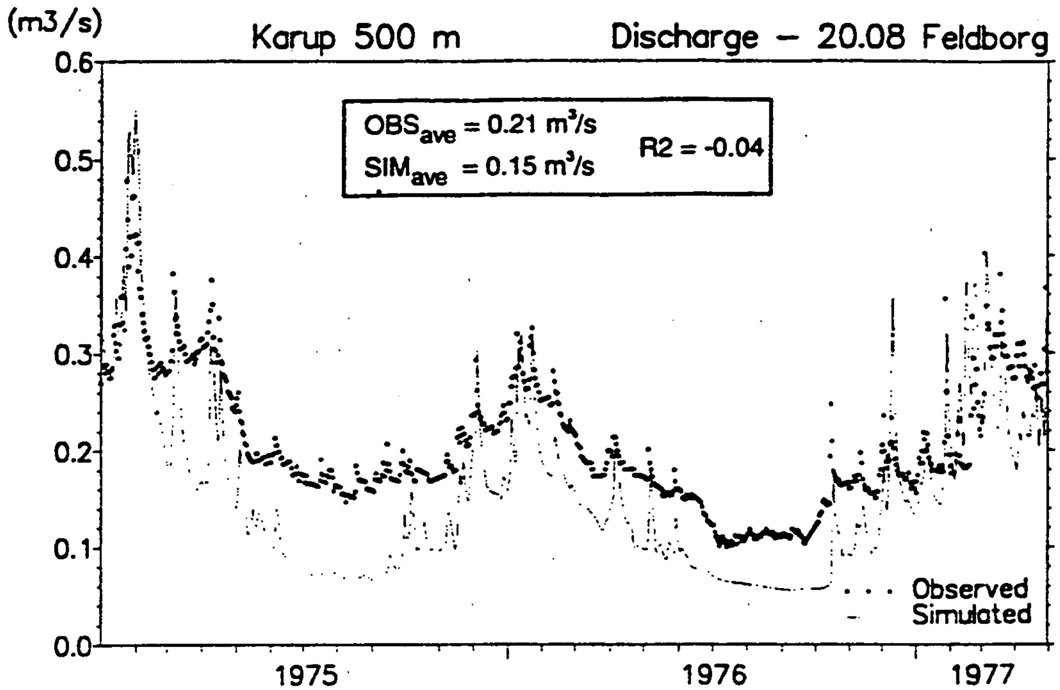


Fig. 8. Continued.



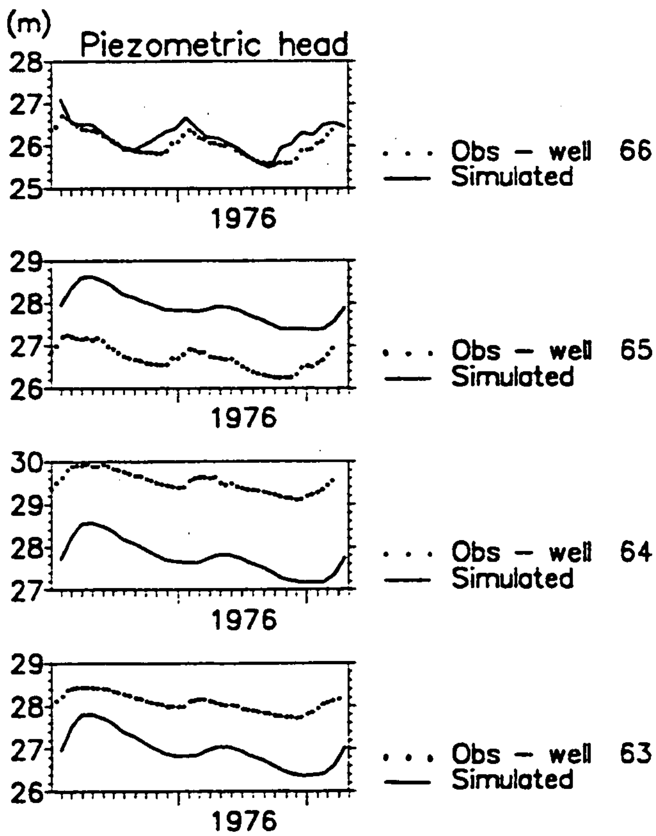


Fig. 9. Simulated and observed piezometric heads from the validation period for four well-sites for which n
calibrations have been made. The locations of the wells are shown in Fig. 2.

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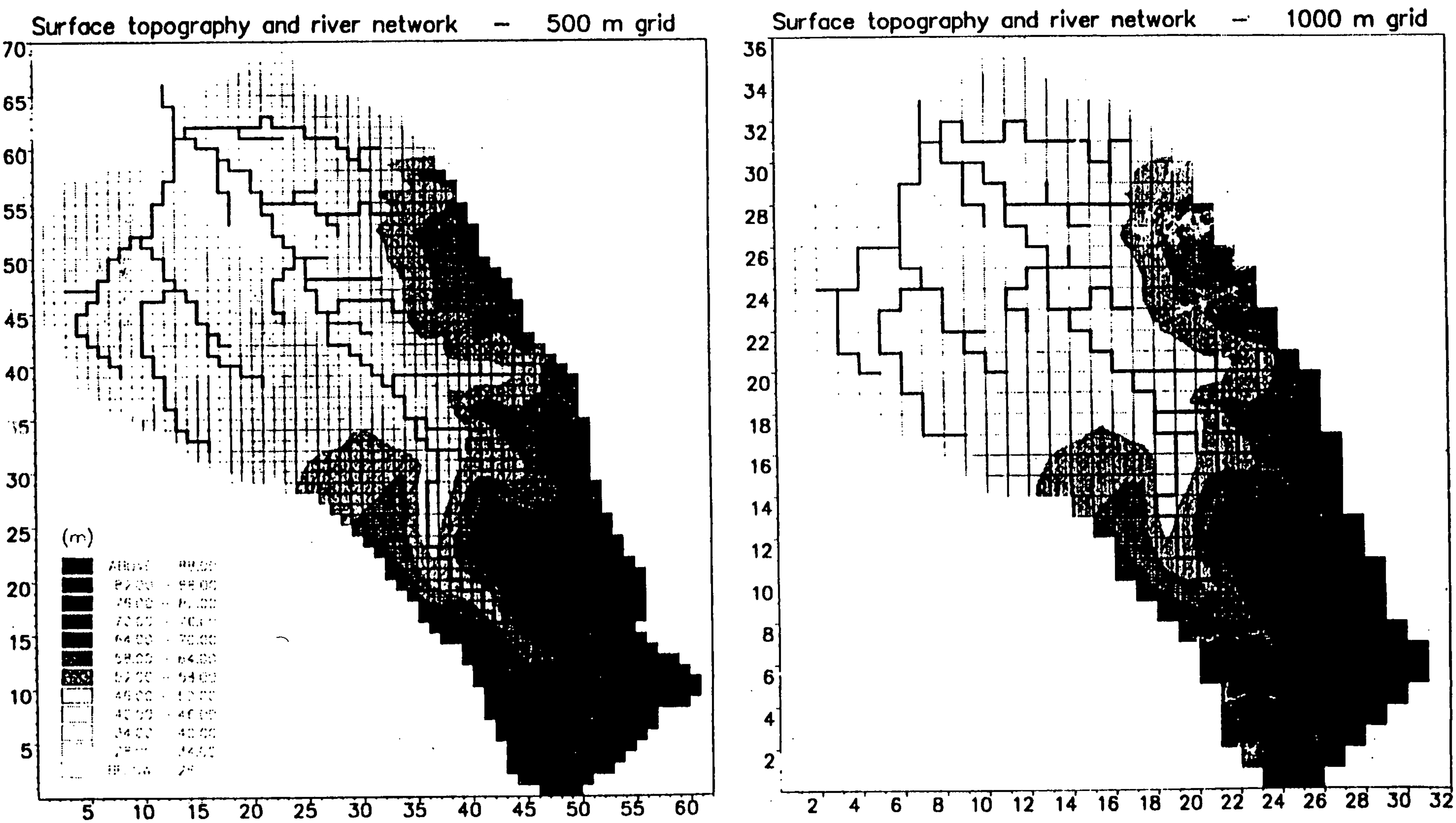


Fig. 10. Topography, river network and model grid for the four models with discretizations of 500, 1000, 2000 and 4000 m.

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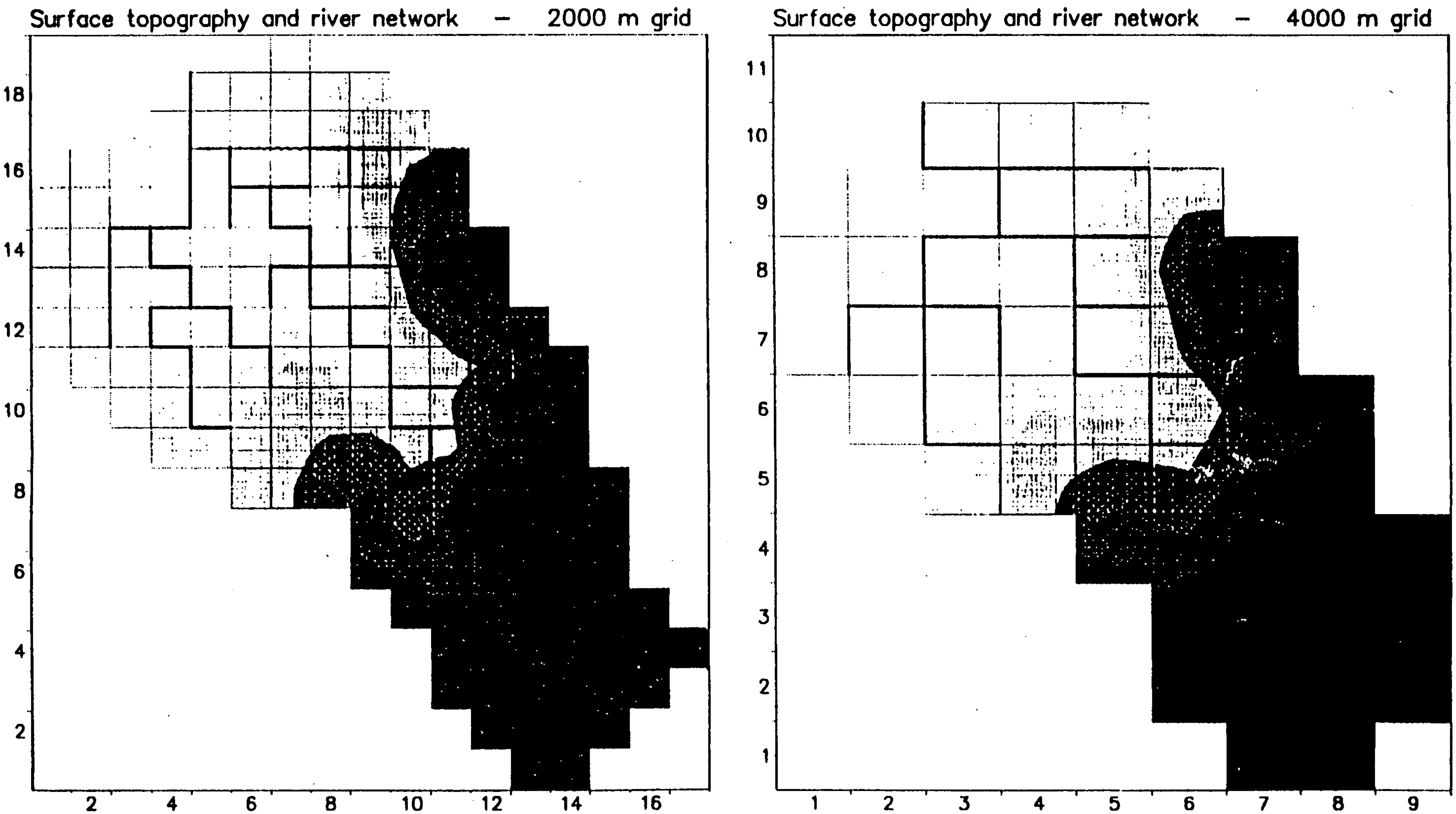


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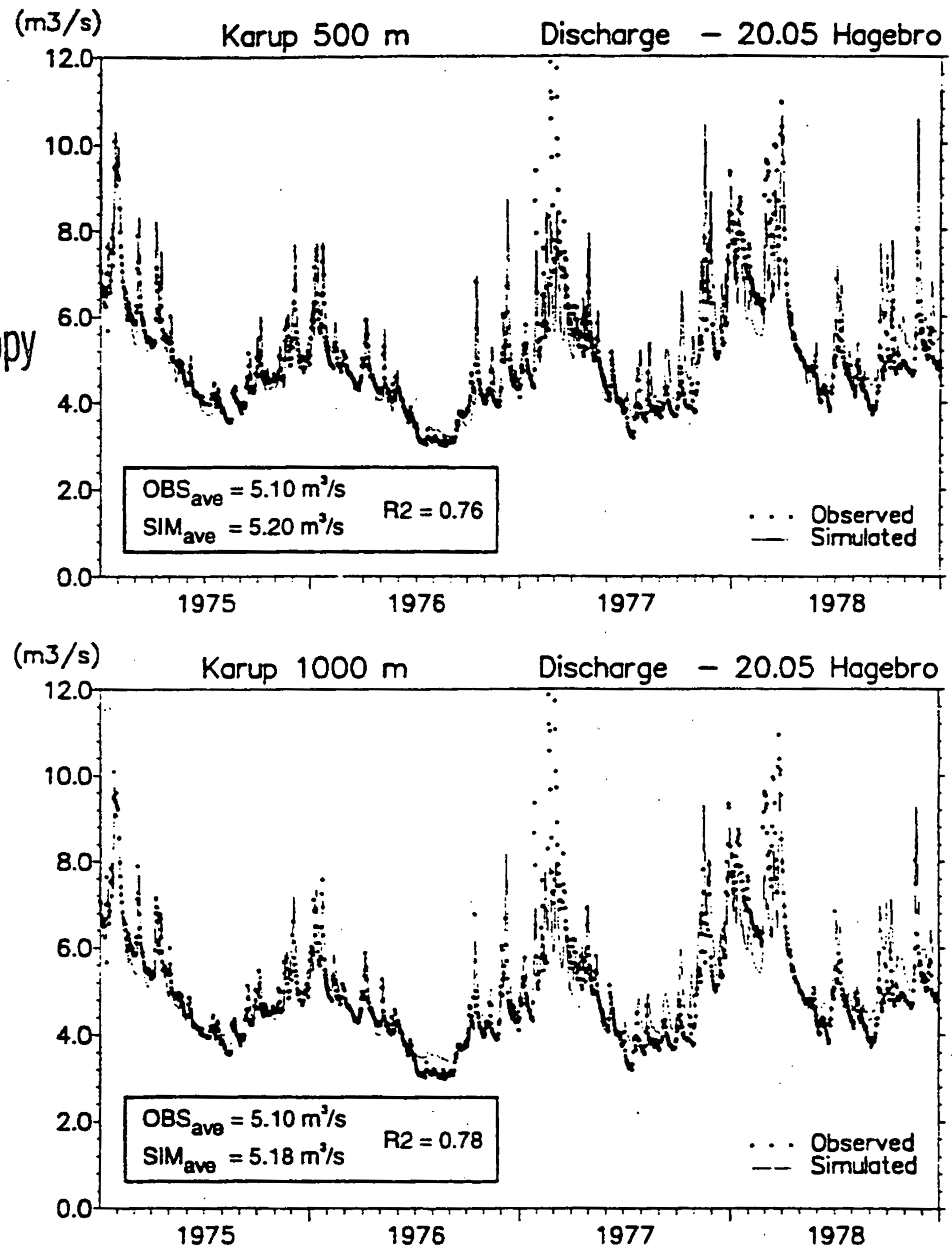


Fig. 11. Simulated and observed discharges average flows, $SIM_{ave}$ and $OBS_{ave}$, and model efficiencies on a daily basis, R2, for the entire catchment using four models with different discretizations.

## 4. Discussion and conclusions

### *4.1. Results and methodology from case study*

The aims of the present paper have been to illustrate a systematic procedure for using a distributed hydrological model and to focus on some of the problems involved. The description of the procedure in the above case study emphasized the importance of the two steps: "(2) establishment of a conceptual model" and "(5) model construction and

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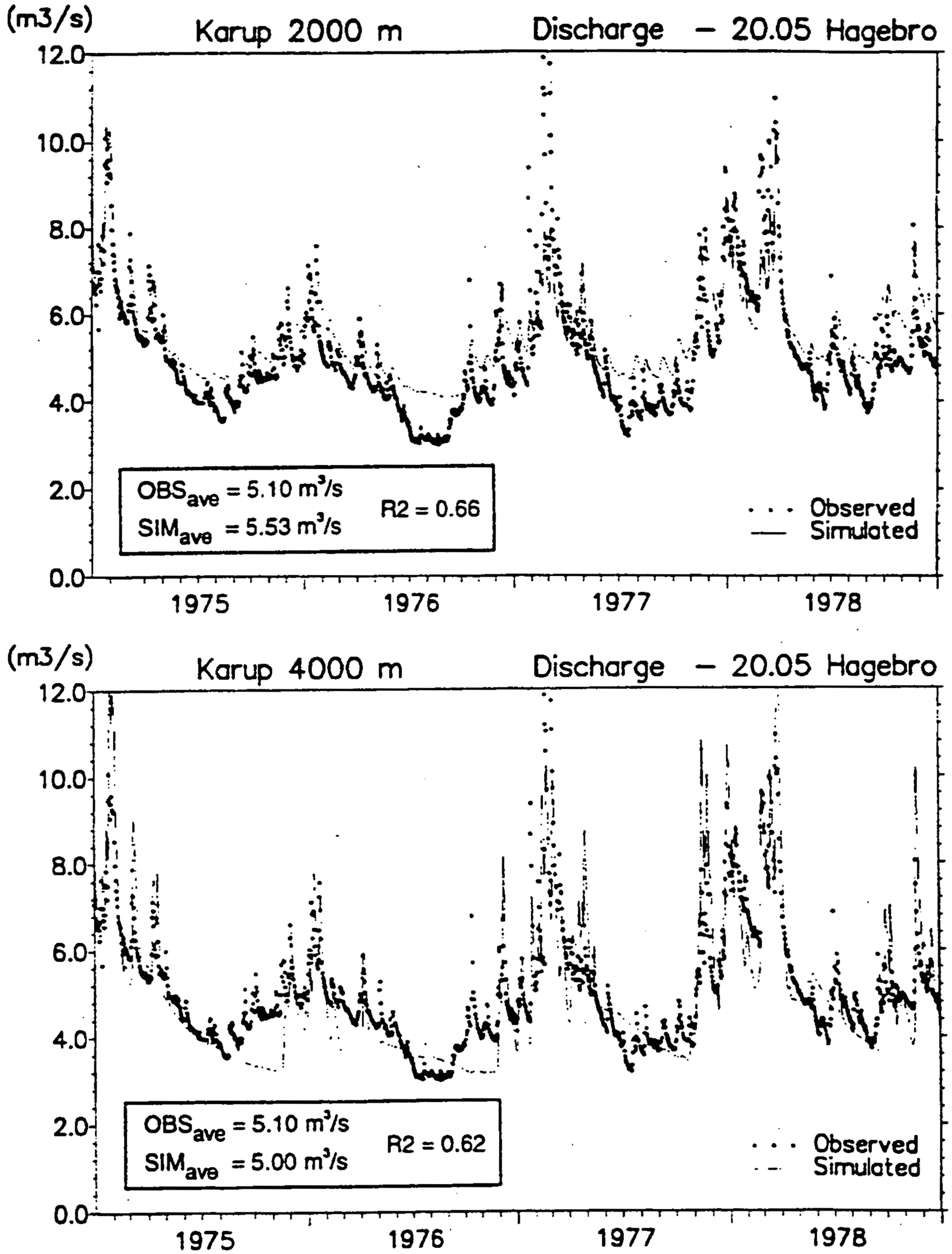


Fig. 11. Continued.

parameterisation''. If these steps are not carried out in a rigorous and purposeful manner, the subsequent steps of model calibration and validation may become rather troublesome. In particular, it is of utmost importance to make a careful parameterisation linked to the specific hydrological conditions and data availability. Experiences from other studies (Refsgaard and Storm, 1996) indicate that an important aspect in this regard is to ensure that the number of parameters subject to adjustments during subsequent calibration becomes as small as possible.

In the above case study, the number of parameters which were subject to calibration was limited to 11. All the other thousands of parameter values were assessed either directly from field data or indirectly through experience from values in similar studies in other catchments. The 11 calibrated parameters comprised three related to aquifer properties and stream–aquifer interaction and eight soil hydraulic parameters (two for each of four different soil types, Table 1). For simulation of discharge and ground water levels it is likely, although not substantiated through calculations, that it would have been possible to limit the eight soil hydraulic parameters to two parameters subject to calibration and six values which were linked by fixed ratios to the ''free parameters''. In this way, the degrees of freedom (''free parameters'') would reduce to five, which is about the same as would be required to calibrate a lumped conceptual rainfall-runoff model to such catchment. Thus, with a rigorous parameterisation approach as adopted in the present case study the ''problem of overparameterisation...'' emphasized by Beven (1996) can be avoided.

More data could have been used for calibration. This would imply that more parameters would have been subject to fitting through calibration. As an example, water level data from the river system could have been used for calibration of Manning numbers. In the present case study, possible errors in the estimated Manning number have, to some extent, been compensated for, as far as the effects on stream–aquifer interaction are concerned, in fitting of the river bed leakage coefficient. This implies that the calibrated parameter loses some of its direct physical interpretation. In general, it may be expected that the more field data that are explicitly utilized the more reliable and the more physically realistic the fitted parameters become. When more independent field data are utilized in the calibration process, the number of parameters to be adjusted through calibration will inevitably increase, but, through a careful parameterisation process, it will still be possible (as in the present case) to keep the ratio between the number of free parameters and the amount of independent field data at a reasonably low level, so that the model does not become overparameterised.

It may be argued that one universal value of hydraulic conductivity for the entire aquifer (the lenses with different conductivities were relatively small and hence of limited importance) is an unusually large simplification for a ground-water model simulating piezometric heads. In this case, the spatially varying thickness of the aquifer is described explicitly, and hence the transmissivity varies significantly even if the hydraulic conductivity is constant. From the validation results (Figs 6 and 7) it appears that one adjustable conductivity value is sufficient for simulating catchment discharge and piezometric head variations at the seven selected observation wells. The limitation of having no spatial hydraulic conductivity variation turned up in connection with the internal validation (Figs 8 and 9). A more refined calibration of aquifer conductivities allowing spatial variations (and hence, more parameter values to adjust) could easily have been done. In fact, such detailed calibration of transmissivities in a ground-water model for the same catchment was done by Miljøstyrelsen (1983), where more than 100 different values were assessed through calibration.

This illustrates that calibration of a distributed model could be a never-ending story, where there are always possibilities for improvements. In this context, it must be noted that the more parameters values that are adjusted through calibration the more field data are required and the more work is required by the modeller. In accordance with Schlesinger et al. (1979) and Klemes (1986) a model should be validated for the types of applications for which it is intended. Thus, performance criteria as well as calibration and validation

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Table 3

An illustration of the need for the incorporation of multicriteria and multi-scale aspects in methodologies for the validation of distributed models

| | Lumped conceptual | Distributed physically based |
|---|---|---|
| Output | At one point:<br>* runoff<br><br>⇒ Single variable | At many points:<br>* runoff<br>* surface water level<br>* ground water head<br>* soil moisture<br><br>⇒ Multi variable |
| Success criteria (excl problem of selecting which statistical criteria to use) | Measured/simulated:<br>* runoff, one site<br><br>⇒ Single criteria | Measured/simulated<br>* Runoff, multi sites<br>* Water levels, multi sites<br>* Groundwater heads, multi sites<br>* Soil moisture, multi sites<br><br>⇒ Multi criteria |
| Typical model application | Rainfall-runoff:<br>* stationary conditions<br>* calibration data exist | Rainfall-runoff, unsaturated zone, ground-water, basis for subsequent water qualitymodellingImpacts of man's activity<br>* non-stationary conditions sometimes<br>* calibration data do not always exist |
| Validation test | Usually "split-sample test" is sufficient<br><br>⇒ Well defined practice exist | More advanced tests required:<br>* differential split sample test<br>* proxy basin test<br><br>⇒ Need for rigorous methodology |
| Modelling scale | Model: catchment scale<br>Field data: catchment scale<br><br>⇒ Single scale | Model: depends on discretization<br>Field data: many different scales<br><br>⇒ Multi-scale problems |

schemes should be tailored to the objectives of the study, and, in general, calibration of a distributed model should not be made more detailed and with more degrees of freedom than is strictly required by the objectives of the modelling study concerned. Hence, also taking into account the user's resources, the most adequate model to be constructed for a specific modelling study is not always the one which explicitly uses all the field data which the selected model code can make use of.

In the multiple scale study with four different model grids (500, 1000, 2000 and 4000 m) the same basic parameter values were used for all four models, although automatic interpolations from the 500 m grid were made in the models with coarser grids. The results (Fig. 11) imply that results identical to those for the 500 m model would have been achieved by discretizations finer than 500 m. The indication that the maximum grid size required for simulation of the catchment discharge is about 1000 m is not surprising when

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considering the topographical variation and the density of the streams in the Karup catchment (Fig. 10). It must be emphasized, however, that for simulation of variables other than catchment discharge, different grid resolutions may be required.

For model discretizations coarser than 1000 m, the results (Fig. 11) indicate that some important processes become scale dependent. This does not necessarily mean that the catchment discharge cannot be simulated by a model with a coarser grid. However, use of a coarser grid would require a significant recalibration and possibly a reformulation of some of the process descriptions to incorporate some of the spatial variability (of topography and stream–aquifer interaction) which are not described explicitly in the coarse grid models.

*4.2. Different validation requirements for lumped and distributed models*

The validation procedure is basically the same for lumped and distributed model codes, but because of the differences in model structures, modes of operation and objectives of application, the validation requirements are much more comprehensive for distributed models. Some of the key differences, summarized in Table 3, are discussed below.

As shown by, e.g. Michaud and Sorooshian (1994) and Refsgaard and Knudsen (1996), lumped models, in many cases, perform just as well as distributed models with regard to rainfall-runoff simulations when sufficient calibration data exist. Therefore, the typical applications for distributed models are, in practice, cases where the modelling requirements extend beyond runoff prediction from gauged catchments, e.g. prediction of runoff from ungauged catchments, water quality simulations and predictions of the effects of changes in land use. This implies that the split-sample validation test traditionally used for lumped models is not sufficient for most distributed model applications.

In connection with the calibration and validation, a success criterion needs to be fulfilled for each output variable for which it is intended to make predictions. Hence, multi-site calibration/validation is needed if spatially distributed predictions are required, and multi-variable checks are required if predictions of the behaviour of individual sub-systems within the catchments are needed. Furthermore, it should be emphasized that, with the present generation of distributed model codes, which do not contain adequate up- or down-scaling methodologies, separate calibration and validation tests have to be carried out every time the grid size is changed.

As shown also in the case study, a model should only be assumed to be valid with respect to outputs that have been explicitly validated. This means, for instance, that a model which is validated against catchment runoff cannot automatically be assumed to be valid also for simulation of erosion on a hillslope within the catchment, because smaller scale processes may dominate here; it will need validation against hillslope soil erosion data.

Another important issue, which has not been addressed by the case study, is related to non-stationarities in catchment conditions, such as predictions of effects of ground-water abstraction or changes in land use. Some authors, e.g. Abbott et al. (1986a) and Bathurst and O'Connell (1992), advocate that distributed models have key advantages as compared to lumped models in this respect. This view is questioned by other authors, e.g. Bergström (1991) and Grayson et al. (1992b), who argue that, at least with the present level of

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available field data, distributed models are not realistic tools for practical water management purposes. No matter whether applications are made for research purposes or in an operational mode it is important to emphasize that non-stationarities create special requirements for validation tests. The only rigorous methodology reported in literature in this regard is a hierarchial scheme of validation tests suggested by Klemes (1986) according to which a so-called differential split-sample test would be required in such case. The basic idea in a differential split-sample test is that the model code should demonstrate an ability to perform through the required transition regime. As this test most often cannot be made on the catchment for which the model is ultimately going to be applied, e.g. for predictions of the effects of future changes in land use, data from similar catchments which have already undergone similar non-stationarities have to be used.

A differential split-sample test is, from a theoretical point of view, weaker than the traditional split-sample test, where data from the specific catchment are used. In connection with validation tests, the uncertainties of the model predictions can be assessed, and a model validity can only be claimed to correspond to a given uncertainty level. In this respect, it is, thus, expected that differential split-sample tests will most often be associated with a higher degree of uncertainty than split-sample tests. However, given that an ordinary split-sample test is not sufficient, conduction of a rigorous and comprehensive differential split-sample test as proposed by Klemes (1986) can be considered as probably the best possible approach.

Although the above discussion focuses on the complications with respect to applying distributed models, this does not imply that they are not suitable for complicated tasks such as prediction of effects of land-use changes. On the contrary, they are most likely the best tools presently available in this respect. However, an important conclusion of the above rigorous validation requirements is that comprehensive validation procedures specifically adapted for each particular application of a distributed model should be used and that statements of their validation status and corresponding predictive capability should, on a case by case basis, be made with more care than has most often been seen until now.

## 5. Acknowledgements

The author wishes to thank Børge Storm, Anders Refsgaard and other colleagues at the Danish Hydraulic Institute for assistance in the model work and for many inspiring discussions over several years which have greatly contributed to the crystallization of the ideas presented in the present paper. The financial support from the Danish Technical Research Council is acknowledged. The constructive criticism and detailed suggestions of the two referees, Marios Sophocleous and Rodger Grayson, are greatly appreciated.

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# Safeguarding the Kristianstad Plain groundwater resource by using the MIKE SHE model

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ABSTRACT: This paper presents the application of a 3D distributed and physically based model, MIKE SHE, for safeguarding the Kristianstad Plain groundwater resource. With the Kristianstad Plain probably being the largest groundwater resource in northern Europe a plan has been needed for future protection. The protection of such a unique and invaluable resource implies an increased understanding of the geohydrological processes, particularly the long term effects of the human activities today and tomorrow. The major questions were: What are the limitations for future abstractions? How will long term threats, like leakage from agricultural land, impact from urban areas and waste deposit sites, affect the excellent groundwater quality of today? MIKE SHE has shown to be a very capable tool to enlighten these kinds of questions.

## 1 INTRODUCTION

The Kristianstad Plain is probably the largest groundwater resource in northern Europe. Its groundwater basin is formed in a depression of the crystalline basement filled up with Cretaceous sedimentary rock, covering an area of approximately 900 sq km. The Cretaceous sedimentary deposits are often more than 150 m thick in the central areas of the Plain and at some places more than 300 m thick. The lower strata is mostly made up of poorly consolidated glauconitic sandstone, which is covered by sandy limestone. The Quaternary deposits are usually 10-20 m thick but depths of more than 60 m have been observed.

The glauconitic sandstone is the major aquifer but also wells placed in limestone and in glaucofluvial deposits give considerable amounts of water. With the Kristianstad Plain being the breadbasket of Sweden the groundwater is mainly used for irrigation, municipal water supply and industries (one is actually the world famous Absolut distillery). The total extraction is approximately 20.5 million m3/year (irrigation 8 million, municipalities 8.5 million, industries 4 million). The yearly groundwater recharge is estimated to 85 million m3/year as an average.

Different extensive studies, concerning the groundwater basin of the Kristianstad Plain, have been carried out by the Swedish Geological Survey, a number of consulting engineering companies, universities, the regional state environmental authority and municipalities during the last six decades. This, of course, makes this area very well documented and there is good knowledge in the geological, hydrological, hydraulic and water quality conditions of the Plain.

The discharged groundwater from the Kristianstad Plain has excellent quality, but there are some severe long term threats, due to leakage from agricultural land, impact from urban areas including waste disposal sites, combined with locally quite large discharges. To be able to protect this unique and invaluable groundwater resource, there is still a need to increase the understanding of the geohydrological processes, particularly the long term effects of land use activities of today and tomorrow. This paper will focus on a study, still in progress, where the MIKE SHE model was applied to achieve this. The study was started in August 1995 and is planned to be completed in the fall of 1997.

The primary issue which is to be solved within the project concerns safeguarding the municipal supply of drinking water for the town of Kristianstad. The Kristianstad water supply station



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provides about 30.000 inhabitants, a number of large food industries and the regional hospital with drinking water. The drilled wells are 70-100 m deep. They are situated very close to the centre of the town and also very close to the municipal waste disposal site, still in use. The risk for groundwater contamination due to this situation is obvious. A new well field needs to be established somewhere else on the Kristianstad Plain, where there are no contradicting interests and no or very small risk that a similar situation as today will occur. The MIKE SHE model was chosen to be the decision tool for this work and forms a base of the entire project.

## 2 BRIEF MIKE SHE DESCRIPTION

MIKE SHE is a deterministic, distributed and physically based modelling system for simulation of geohydrological processes in the entire land phase of the hydrological cycle in a given catchment. The following components are encountered in the system: interception-evapotranspiration, overland-channel flow, unsaturated flow, groundwater flow, aquifer-river exchange, snow melt.

The catchment is discretized in the horizontal plane by a grid square network which is used both in the overland flow component and the groundwater flow component. These are linked by a vertical column of nodes at each grid representing the unsaturated zone. Water movements in the catchment are modelled by a numerical solution (finite difference) of the partial differential equations describing the processes of overland and channel flow, unsaturated and saturated subsurface flow, interception, evapotranspiration and snow melt.

The model is applicable to a wide range of water resources and environmental problems related to surface water and groundwater systems and the dynamic interaction between these. MIKE SHE has shown to be a very capable tool when the effects of human interference, e.g. pollutant loading from waste disposal sites, changes in land use, is to be assessed. A detailed description of the MIKE SHE model is presented in DHI (1993) and DHI (1995).

## 3 THE KRISTIANSTAD PLAIN MODEL

The entire catchment area is approximately 2 400 sq km limited by the natural watershed. It contains the entire area of the Kristianstad Municipality but also parts of three other municipalities, Hässleholm, Bromölla and Sölvesborg.

### 3.1 *Input data*

A large amount of information has been collected, put together and digitised into the MIKE SHE model.

The model limit was chosen to coincide with the watershed to simplify the setting of the boundary conditions The model area only cuts the watershed in two discrete points (the rivers Helgeån and Skräbeån). The boundaries in these two points were set to measured time series of the river flows.

The topography was described with contour lines from the national topographical map with 5 meters level interval. The surface characteristics were described by parameters for roughness, distribution codes for landuse and crops. The main rivers through the catchment were described with cross-sections, bank levels, roughness and river bed lining characteristics.

The geological model has been built with a typical precision of 200 meters. In total 8 geological layers were defined, starting from top with Peat, Sand, Clay, Gravel, Till, Limestone, Sandstone, and finally as bottom layer, the Bedrock. See example of cross-section in Figure 1. The upper soil characteristics (i.e. porosity and hydraulic conductivity) were described for each type of soil profile relevant for the unsaturated zone. The hydraulic parameters for the saturated zone were among others estimated from a large number of pumping experiments, later on verified through model runs. The saturated zone was discretized by five vertical simulation layers, more or less corresponding to the major geological layers.

The actual water abstractions from the different aquifers, in total more than 600 municipal, industrial, private and irrigation discharges, were described as time series for the chosen simulation period, 1980 to 1990. In addition, a number of larger embankments were described, being the largest source of discharge from the top layers.

Finally, a number of climatological variables were described by continuous time series, covering 1980 to 1990. The daily precipitation were given as input from fifteen stations, distributed according to the typical annual precipitation. Temperature were taken from two stations representing the typical



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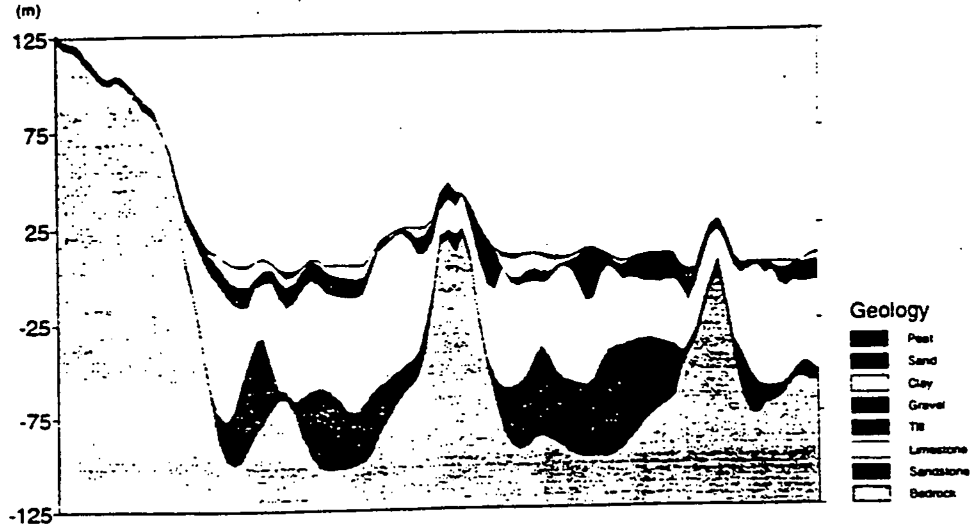


Figure 1. Cross section showing the geological layers through the town of Kristianstad.

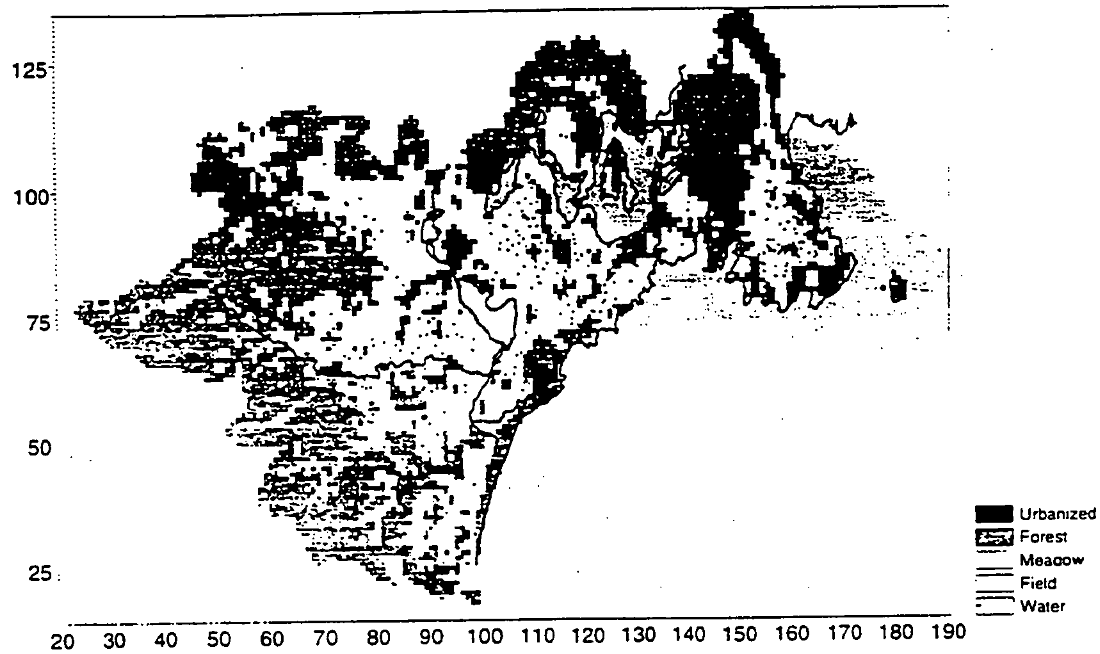


Figure 2. The figure shows the landuse for the entire area modelled. The thin line is representing rivers and coastal lines. The small dots indicate the major abstraction wells.

difference between the lower Plain and the ridges. In addition. the evaporative demand was described by typical monthly values of potential evaporation from four weather stations. and leaf area index and root depth for the different types of vegetation and crops within the area.

3.2 *Verification of the model*

Hydraulic parameter values were initially set to values based on overall experience and a large number of pumping tests carried out during the last decades. Generally, the initial values for the various layers, were given as lumped values for large sub-areas. By comparing the simulated



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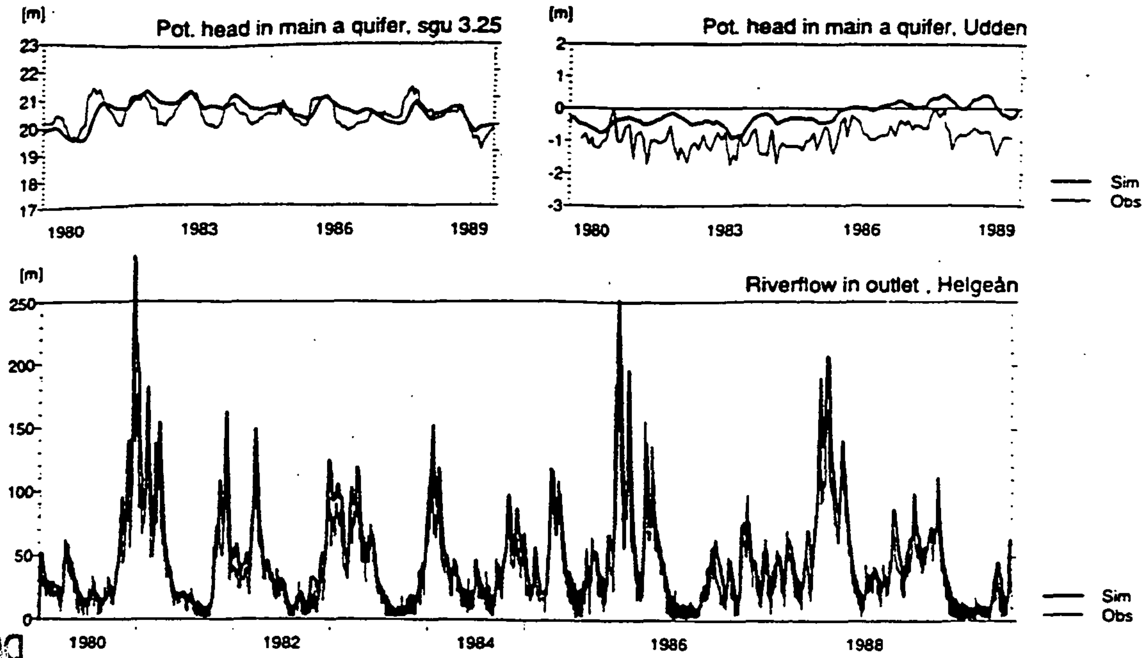




Figure 3. Example of verification results at two important groundwater observation sites in the main aquifer, and the main river outlet from the watershed - Helgeån.

hydraulic head pattern with observed ones, the hydraulic parameters were adjusted until satisfactory comparison was achieved. About 50 different groundwater level observation sites were used to verify the groundwater model.

In addition, a number of river flow stations were used to verify the overall water balance, as well as the exchange flow between the aquifers and the river network. The water balance shows that approximately 14% of the yearly precipitation leaves the catchment area through rivers and more than 70% evaporates. This leaves 16% for infiltration of which only 2% percolates all the way down to the limestone and sandstone.

The model is at this stage verified with a grid size of 1000 by 1000 meters. Some examples are shown in Figure 3. The process of validating the model results with a grid size of 500 by 500 meters is right now in progress. So far, the results look as promising as for the 1000-grid model. The work with a number of sub-models with finer grid size (e.g. 250 and 50 meters) are in progress. These will be verified accordingly.

## 4 MODEL APPLICATIONS

Many different scenarios can be, have been and are to be simulated, based on both historical events and future expected and exceptional conditions. In especially interesting areas, sub-models will be created using results from the entire model as boundaries.

One sub-model, covering 200 sq km, has already been established for more detailed studies of the area around the town of Kristianstad. The grid size for this model was set to 250 by 250 meters, and the applications of the model range from detailed hydraulic analyses to overall risk analyses from different sources of contamination. A sub-model with a grid size of as little as 50 by 50 meters, dedicated for detailed transport studies of the Kristianstad waste disposal site, is right now being built. For this model a much finer raw data precision is used, especially for topography and geology. The limits of the sub-models already established or in progress, as well as some landmarks and the main aquifer, are illustrated in Figure 4.

The scenarios studied this far, except for present conditions (verification), comprise:

- Natural conditions, meaning no human interference. Among others, the simulation results show that, without any extraction from the aquifer, the potential head would be just below the ground level in the centre of Kristianstad. These results have been verified against information from the beginning of this century (before the extensive discharges started). At present conditions the groundwater depletion is approximately 5 meters.
- Alternative locations of a new well field for the Kristianstad municipal water. Three alternative locations have been briefly analysed. The most interesting alternatives will be studied closer in a more detailed sub-model.

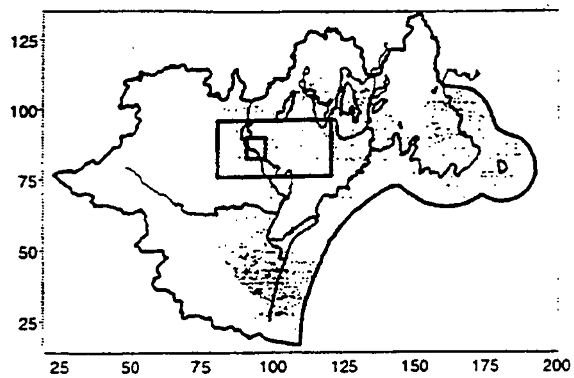


Figure 4. Example of sub-models (rectangles) with finer grid size. The grey area indicates the limits of the main aquifer.

In addition to these water movement studies, a number of contamination scenarios are interesting to study. These simulations will be carried out with a pure advection/dispersion approach, which simulates the detailed transport and spreading of dissolved conservative solutes. In the first step these simulations will only be applied on the saturated zone. Additional input data for these simulations include: effective porosity, longitudinal and transversal dispersion parameters, and the concentration of the contaminating source. Studies already started or planned comprise:

- Risk analysis of the municipal waste disposal site contaminating the groundwater wells within the town limits of Kristianstad. An overall analysis with the 250-grid model has already been carried out. The results from this study can however only be used as a very rough indication of the contamination risk. The final risk analyses will be carried out with a much more precise description, i.e. the 50-grid model. Only to illustrate possible results from such simulations, the transport of pollutants from the waste disposal site is shown in Figure 5 and Figure 6.
- Risk analysis of contamination from tank truck accidents.
- Mapping of the age of the water at different locations in the aquifer. This interesting analyses will be managed by a newly developed add-on module to MIKE SHE, Refsgaard et al (1995).

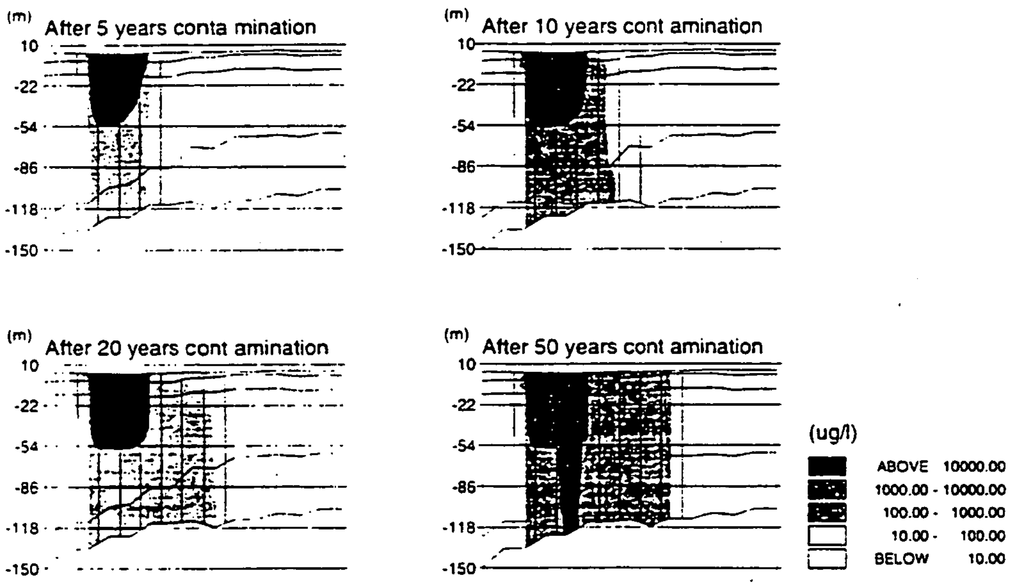


Figure 5. Transport and spreading of the contamination from the waste disposal site shown as a cross-section (south-east to north-west) through the waste disposal site and the centre of the well field.



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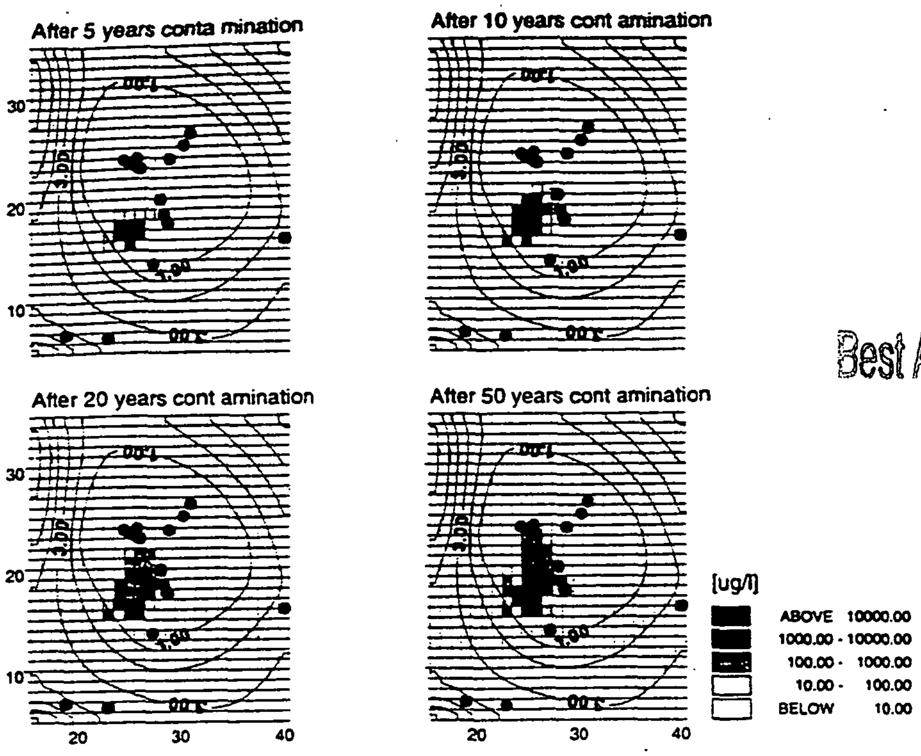




Figure 6. Transport and spreading of the contamination in the main aquifer from the waste disposal site close to the centre of Kristianstad (based on indicative simulations with a 250-grid sub-model).

## 5 CONCLUSIONS

Integrated mathematical models describing the interaction between surface and subsurface water systems in a physically correct way are valuable tools when analysing the effects of human interference on the hydrological cycle.

The presented work, which only constitute the first step of the planned studies on the Kristianstad Plain, have clearly demonstrated that MIKE SHE successfully can be applied for analysing regional surface water bodies and sub-surface groundwater systems, as well as the interaction between the two.

Through the establishment of the geological model and the calibration of the hydrological model a great knowledge about the geometry of the sub-surface environment was obtained and stored in the database of the modelling system. Due to the flexibility of the database, it is easy to improve the geological interpretation through new borings, etc., and consequently easy to incorporate changes in the hydrological model in the future.

With the verified model, some initial successful studies have been carried out to evaluate alternative well fields, and in general increase the understanding of the sub-aquifers and their interaction.

The three-dimensional modelling of contaminant transport in the groundwater due to waste disposal sites has been illustrated by use of a combined regional and local model approach, where time-varying boundary conditions for a local model with fine discretizations are generated by a regional (flow) model with a more coarse discretization.

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# Water management of the Gabcikovo Scheme for balancing the interest of hydropower and environment

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ABSTRACT
The Gabcikovo hydropower scheme on the Danube is located in the area comprising one of Europe's most unique riverine wetland ecosystems and one of Central Europe's largest ground water resources which receives the main part of its recharge from infiltration through the river bed. When Gabcikovo was put into operation by Slovakia in 1992, and the main part of the water was diverted away from the Danube and the associated river branch system over a stretch of approximately 30 km and a reservoir was created on the Danube, it obviously was a matter of serious concern requiring comprehensive environmental assessment studies. This paper describes a comprehensive modelling system integrating modules for describing the reservoir (2D flow, eutrophication, sediment transport), the river and river branches (1D flow including effects of hydraulic control structures, water quality, sediment transport), the ground water (3D flow, solute transport, advanced geochemistry), agricultural aspects (crop yield, irrigation, nitrogen leaching) and flood plain conditions (dynamics of inundation pattern, ground water and soil moisture conditions, and water quality). A few selected results from the model applications are shown.

## 1 INTRODUCTION

The Danubian Lowland (Figure 1) between Bratislava and Komárno is an inland delta formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which throughout the year receives in the order of 30 $m^3/s$ infiltration water from the Danube in the upper parts of the area and returns it into the Danube and the drainage channels in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

Human influence has gradually changed the hydrological regime in the area. Construction of dams upstream of Bratislava together with exploitation of river sediments has significantly deepened the river bed and lowered the water level in the river. These changes have had a significant influence on the conditions of the ground water regime as well as the sensitive riverside forests downstream of Bratislava. In spite of this basically negative trend the river branch system with its alluvial forests and the associated wetland ecosystems still represents a very unique landscape of outstanding importance.

The Gabcikovo hydropower scheme was put into operation in 1992. A large number of hydraulic structures has been established as part of the hydropower scheme. The key elements are a system of weirs across the Danube at Cunovo 15 km downstream of Bratislava, a reservoir created by the damming at Cunovo, a new lined canal running parallel to the Old Danube over a stretch of approximately 30 km for navigation and with intake to the hydropower plant, a hydropower plant and two shiplocks at Gabcikovo, and an intake structure at Dobrohost diverting water from the new canal to the Slovak river branch system.

The entire scheme has significantly affected the hydrological regime and the ecosystem of the region. Thus, before the damming of the Danube in 1992 the river branches were connected with the Danube during periods with discharge above average. However, some of the branches were only active during flood situations a few days per year. After the damming the water level in the Old Danube has decreased significantly. Therefore, in order to avoid that most water drained from the river branches to the Old Danube, resulting in totally dry river branches, the connections between the Danube and the river branches have been blocked except for the downstream one at chainage 1820 rkm (see Figure 2). Instead, the river branch system receives water from the inlet structure in the hydropower canal at



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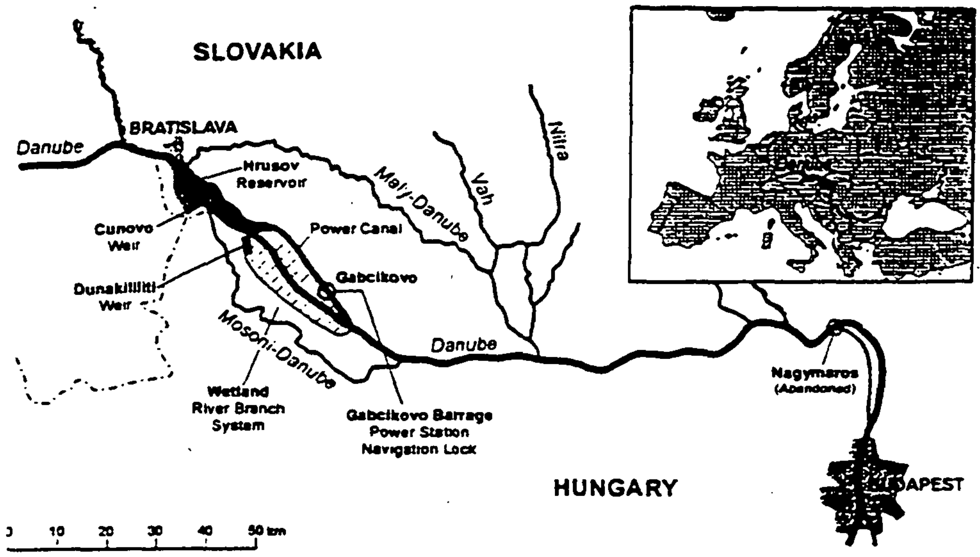


Figure 1. The Danubian Lowland with the new reservoir and the Gabcikovo scheme.

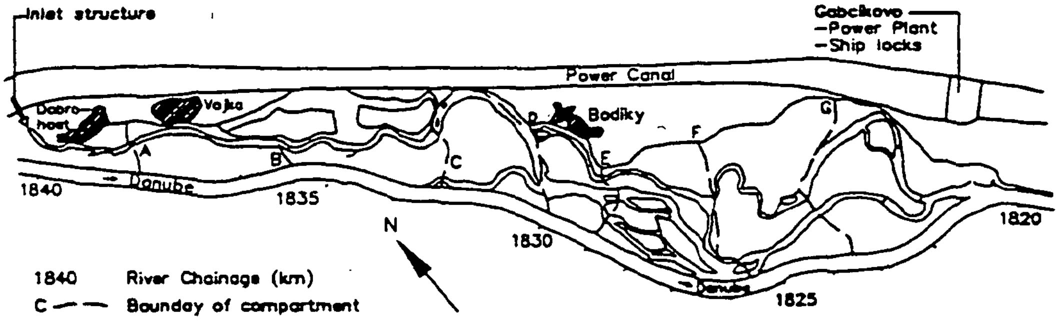


Figure 2. Layout of the river branch system on the Slovakian side of the Danube

Dobrohost (Figure 2). An overview of the monitoring and assessments of environmental impacts is provided in Mucha (1995).

The scheme was originally planned and the major parts of the construction were carried out as a joint effort between Czecho-Slovakia and Hungary in accordance with an international treaty from 1977. However, today Gabcikovo is a matter of controversy between Slovakia and Hungary, who have referred some disputed questions to the International Court of Justice in Haag.

The present paper describes a comprehensive integrated modelling system established for studying the impacts of alternative water management decisions on the environment. Furthermore, a few selected results of model applications are shown.

## 2. INTEGRATED MODELLING SYSTEM

### *2.1. Individual model components*

An integrated modelling system (Figure 3) has been established by combining the following existing and well proven mathematical modelling systems:

- *MIKE SHE* (Refsgaard and Storm, 1995) which, on catchment scale, can simulate the major flow and transport processes of the hydrological cycle:
  - 1-D flow and transport in the unsaturated zone
  - 3-D flow and transport in the ground water zone
  - 2-D flow and transport on the ground surface
  - 1-D flow and transport in the river.

All the above processes are fully coupled allowing for feedback's and interactions between components. In addition, MIKE SHE includes modules for multi-component geochemical and



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biodegradation reactions in the saturated zone (Engesgaard, 1996).

- *MIKE 11* (Havnø et al., 1995), which is a one-dimensional river modelling system. MIKE 11 is used for hydraulics, sediment transport and morphology, and water quality. MIKE 11 is based on the complete dynamic wave formulation of the Saint Venant equations. The modules for sediment transport and morphology are able to deal with cohesive and non-cohesive sediment transport, as well as the accompanying morphological changes of the river bed. The non-cohesive model operates on a number of different grain sizes, taking into account shielding effects.
- *MIKE 21* (DHI, 1995), which has the same basic characteristics as MIKE 11, just extended to two horizontal dimensions, is used for reservoir modelling.
- MIKE 11 and MIKE 21 include *River/Reservoir Water Quality (WQ) and Eutrophication (EU)* (Havnø et al., 1995; VKI, 1995) modules to describe oxygen, ammonium, nitrate and phosphorus concentrations and oxygen demands as well as eutrophication issues such as bio-mass production and degradation.
- *DAISY* (Hansen et al., 1991) is a one-dimensional root zone model for simulation of soil water dynamics, crop growth and nitrogen dynamics for various agricultural management practices and strategies.

### 2.2. *Integration of model components*

The integrated modelling system is formed by the exchange of data and the feed-backs between the individual modelling systems. The structure of the integrated modelling system and the exchange of data between the various modelling systems are illustrated in Figure 3. The interfaces A-E between the various models are briefly described below:

A) MIKE SHE forms the core of the integrated modelling system having interfaces to all the individual modelling systems. The coupling of MIKE SHE and MIKE 11 is a fully dynamic coupling where data is exchanged after each computational time step. The MIKE SHE-MIKE 11 coupling is crucial for a correct description of the dynamics of the river-aquifer interaction. Firstly, the river width is larger than one MIKE SHE grid, in which case the MIKE SHE river-aquifer description is no longer valid. Secondly, the river/reservoir system comprises a large number of hydraulic structures, the operation of which cannot be accounted for in MIKE SHE. Thirdly, the very complex branch system with loops and flood cells needs a very efficient hydrodynamic formulation such as MIKE 11's.

The remaining modelling systems are coupled in a more simple manner involving a sequential execution of individual models and subsequently a transfer of boundary conditions from one model to another. Some examples are listed below.

B) Results of eutrophication simulations with MIKE 21 in the reservoir are used to estimate the concentration of various water quality parameters in the water that enters the Danube downstream of the reservoir to be used for water quality simulations for the Danube using MIKE 11.

C) Sediment transport simulations in the reservoir with MIKE 21 provide information on the amount of fine sediment on the bottom of the reservoir. This information is used to calculate leakage coefficients which are used in ground water modelling with MIKE SHE to calculate the exchange of water between the reservoir and the aquifer.

D) The DAISY model calculates vegetation parameters which are used in MIKE SHE to calculate the actual evapotranspiration. Ground water levels calculated with MIKE SHE act as lower boundary conditions for DAISY unsaturated zone simulations. Consequently, this process is iterative and requires a few model simulations.

E) Results from water quality simulations with MIKE 11 and MIKE 21 are used to estimate the concentration of various species in the water that infiltrates to the aquifer from the Danube and the reservoir. This is being used in the ground water quality simulations (geochemistry) with MIKE SHE.

The Danubian Lowland Information System (DLIS) is a combined data base and geographical information system which has been developed under this project. The DLIS is based on Informix (database) and Arc/Info (GIS) and provides a framework for data storage, maintenance, processing and presentation. In addition, an interface between DLIS and MIKE SHE allowing import and export of maps and time series files in MIKE SHE file formats has been established (Sørensen et al., 1996).

### 2.3. *Comparisons to other modelling systems*

No other modelling systems including the same level of comprehensive physically-based model codes for the various components and a similar level of integration among the different codes have been seen in the literature.



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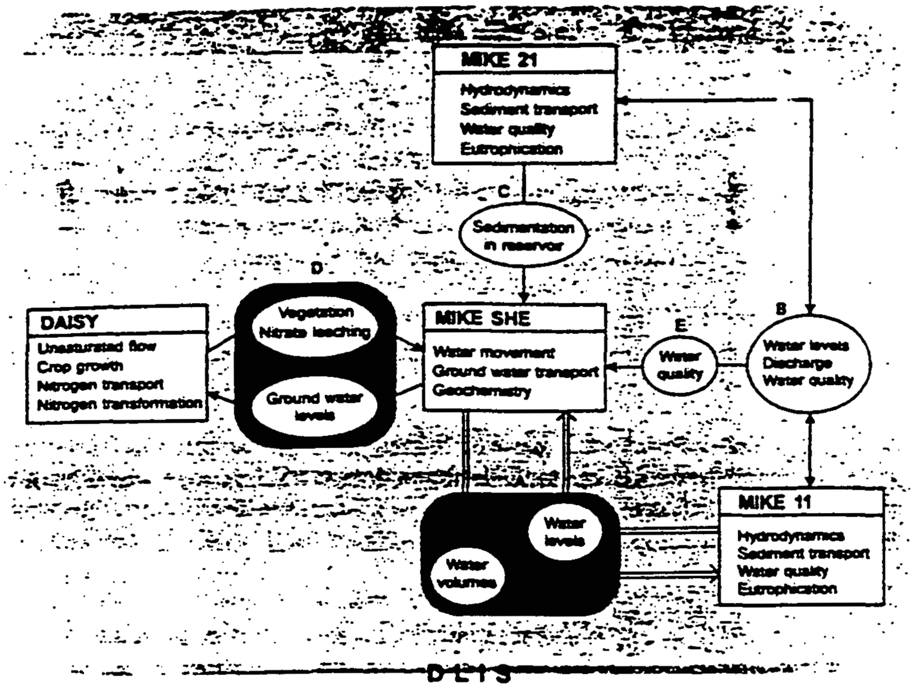




Figure 3. Structure of the integrated modelling system with the key interactions between the individual models.

Yan and Smith (1994) described the demand and outlined a concept for a full integrated ground water - surface water modelling system including descriptions of hydraulic structures and agricultural irrigation as a decision support tool for water resources management in South Florida.

Typical examples of codes described as integrated in the literature are Menetti (1995) and Koncsos et al. (1995). However, they either consist of independent model codes, which are not coupled (Menetti, 1995) or does not include both the surface and the subsurface processes and their interaction (Koncsos et al., 1995)

## 3. MODEL CONSTRUCTION, CALIBRATION AND VALIDATION

All the applied models are based on distributed physically-based model codes. This implies that most of the required data for establishing the model set-up and input data can be measured directly in the field.

Nevertheless, some model calibration was necessary in order to improve the model performance by making use of the information contained in existing data on water levels, flow velocities, ground water tables, sedimentation and erosion as well as concentrations of various solutes in surface and ground water.

After the model calibration, the predictive capability of the models were tested against field data for periods outside the calibration periods. A fundamental issue in this regard is that the aim of the subsequent model applications are in general to predict the environmental impacts of alternative water regimes differing from situations for which data are available for validation tests. Hence, it is not possible to carry out very rigorous tests in all respects, such as ideally proposed by Klemes (1986). However, the validation tests were so comprehensive as possible, taking advantage of the fact that the damming of the Danube and establishment of the reservoir in 1992 has created a new hydrological situation, which in many respects differs fundamentally from the pre-dam situation. Thus, wherever possible the models were calibrated on pre-dam conditions and validated on post-dam conditions. Results from the model calibration and validation tests are described in DHI et al. (1995).

## 4. EXAMPLES OF MODEL APPLICATION

### *4.1 Water management scenarios*

One historical (pre-dam) and three hypothetical water management regimes corresponding to alternative operation rules for the structures of the Gabcikovo



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Table 1. Discharge characteristics (in $m^3/s$) of the four different water management regimes (WMR's) for a five year period corresponding to upstream discharge equivalent to the 1986-91 period

| | WMR I. (Pre-dam) | WMR II (Post-dam) | WMR III (Post-dam) | WMR IV (Post-dam) |
|---|---|---|---|---|
| Average flows | | | | |
| Old Danube, downstream of the reservoir | 2017 | 393 | 732 | 218 |
| Intake to river branch system at Dobrohost | - | 45 | 45 | 37 |
| Gabcikovo hydro power | - | 1469 | 1130 | 1653 |
| Maximum flows | | | | |
| Old Danube, downstream of the reservoir | 9022 | 5598 | 5598 | 5598 |
| Intake to river branch system at Dobrohost | - | 234 | 234 | 234 |
| Gabcikovo hydro power | - | 3080 | 3080 | 3080 |
| Minimum flows | | | | |
| Old Danube, downstream of the reservoir | 745 | 100 | 133 | 50 |
| Intake to river branch system at Dobrohost | - | 7 | 7 | 7 |
| Gabcikovo hydro power | - | 480 | 284 | 578 |

system were simulated. The assumed operation rules for separation of the discharge to the three main 'routes', namely to the Old Danube, through the Gabcikovo hydropower plant and to the Slovak river branch system through the intake structure at Dobrohost, are described in DHI et al. (1995). A summary of the resulting discharge regimes are shown in Table 1.

For each of the three post-dam scenarios, two situations have been considered, namely a situation corresponding to establishment of eight so-called underwater weirs and a situation without such weirs. These weirs, which have a design height of 4-5 m, are totally drowned, but causes an increase of the water levels and thus an improvement of the ecologically vital connectivity between the Old Danube and the river branch system. Thus, a total of seven scenarios has been studied.

In the following a few selected examples of model applications are shown. The examples concern the effects of different water management regimes on water quality in the Old Danube and on hydrological conditions in the river branch system. Further results including aspects relating to the reservoir, sedimentation, ground water quality and crop yields, are described in DHI et al. (1995).

### 4.2 Water quality in the Danube

The water quality in the Old Danube is important both for flora and fauna in the water and for the other water bodies affected by it, such as the ground water system and the Danube further downstream. The following model calculations were carried out for the scenarios:

- Hydraulic calculations of water levels and flow velocities.
- Sedimentation transport, both for fine suspended sediment which can settle in areas with low velocities, and for bed load of coarse sand and gravel material, which may result in long term morphological changes in river cross- and long-sections.
- Water quality in terms of dissolved oxygen concentrations.

The model runs were carried out by MIKE 11 using upstream boundary conditions calculated by other models. For example the MIKE 21 reservoir model provided data on sediment concentrations of different sediment fractions as well as of water quality variables ($NO_3$, $NH_4$ and BOD). In the following only results from the water quality calculations are shown and discussed.

The only situations which seen from a water quality point of view are potentially critical are during the summer period. The most critical location in the Old Danube is the reach just upstream the confluence with the outlet canal from the hydropower plant.

In Figure 4 the minimum oxygen concentrations for such a worst case situation is shown for all the four water management regimes, including the underwater weirs' scenarios for the three post-dam regimes. The results indicate that the oxygen concentrations are of the same order of magnitude for WMR II and III as for the pre-dam condition (WMR I), and WMR III with average 732 $m^3/s$ even appears to result in slightly higher oxygen conditions than the pre-dam situation.

The introduction of underwater weirs increases the water level - and thereby the cross-section area, thus leading to decrease in the flow velocities and a higher



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retention time. Increased water levels and decreased flow . velocities and the reaeration have the consequence that the respiration processes become more important to the oxygen concentration in the river. From Figure 4 it can be seen that this, in general, results in lower oxygen concentrations in the scenarios with underwater weirs than in the scenarios without.

It must be emphasised that the predicted concentrations shown in Figure 4 are minimum concentrations that may occur rarely, when there is coincidence of critical discharges and critical summer periods with high algae content, high temperature and thereby high respiration in the water column. The oxygen concentrations show diurnal variations with maximum values typically 4-5 mg/l higher than the minimum values shown in Figure 4. On this basis it has been concluded that for WMR's I, II and III no water quality problems are foreseen, while the oxygen conditions in WMR IV may become critical, especially for situations with underwater weirs.

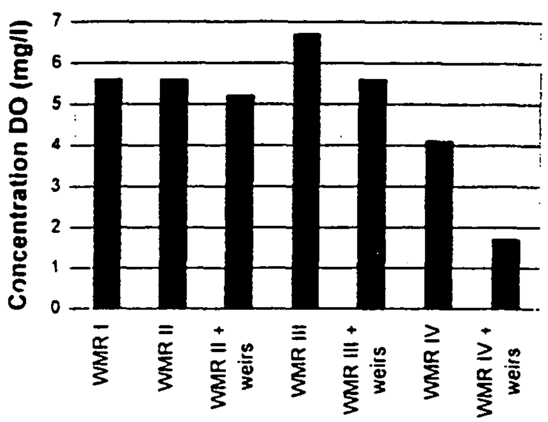


Figure 4. Minimum concentrations of dissolved oxygen in the Old Danube for a worst case situation.

*4.3 Hydrological conditions in the river branch system*

The complexity of the river branch system is indicated in Figure 2 for a 20 km reach downstream the reservoir on the Slovakian side where alluvial forest occurs. In order to enable predictions of possible changes in floodplain ecology it is crucial to provide a detailed description of both the surface water and the groundwater systems in this area as well as of their interaction. For this purpose the fully coupled MIKE SHE-MIKE 11 model was used. The horizontal discretization of the model is 100 m, while the groundwater zone is represented by two layers. The computational time step was approximately one hour. Several hundreds of cross-sections and more than 50 hydraulic structures in the river branch system were included in the MIKE 11 set-up for the river system.

The floodplain model is a management tool which can simulate the operation of the hydraulic structures, enabling an optimisation of the hydraulic and ecological conditions for the unique floodplain environment. The floodplain model provides detailed information in time and space about water levels in river branches and on the floodplains, groundwater levels and soil moisture conditions in the unsaturated zone. Such information can directly be compared with quantitatively formulated ecological criteria.

As an example of the results which can be obtained by the floodplain model Figure 5 shows a characterisation of the area according to flooding and depths to groundwater. The upper map has been processed on the basis of simulations with 1988 discharge data for pre-dam conditions (WMR I). The classes with different ground water depths and flooding have been determined from ecological considerations according to requirements of (semi)terrestrial (floodplain) ecotopes. From the figure the contacts between the main Danube river and the river branch system is clearly seen. Similar computations have been made by two other water management schemes after damming of the Danube (WMR II and WMR III). The results of WMR II without underwater weirs are shown on the lower map in Figure 5. By comparing the two maps the differences in hydrological conditions can clearly be seen. From such changes in hydrological conditions inferences can be made on possible changes in the floodplain ecosystem.

It may be noted that WMR II resembles the operation practise which has been carried out since 1992. Hence, these model results indicate that ecological changes are likely to occur in this area: however the time horizon for such fundamental changes in flora and fauna is decades. Whether such changes should be considered as positive or negative depend on the ecological objectives for this area. In this respect the situation is classic with conflicting interests between different sectors, not only between hydropower and ecology, but also between differing ecological objectives such as forestry, fishery, recreation and nature conservation. The long term nature of the potential ecological changes implies that there is still sufficient time to decide on other operation policies with other impacts, if desired.



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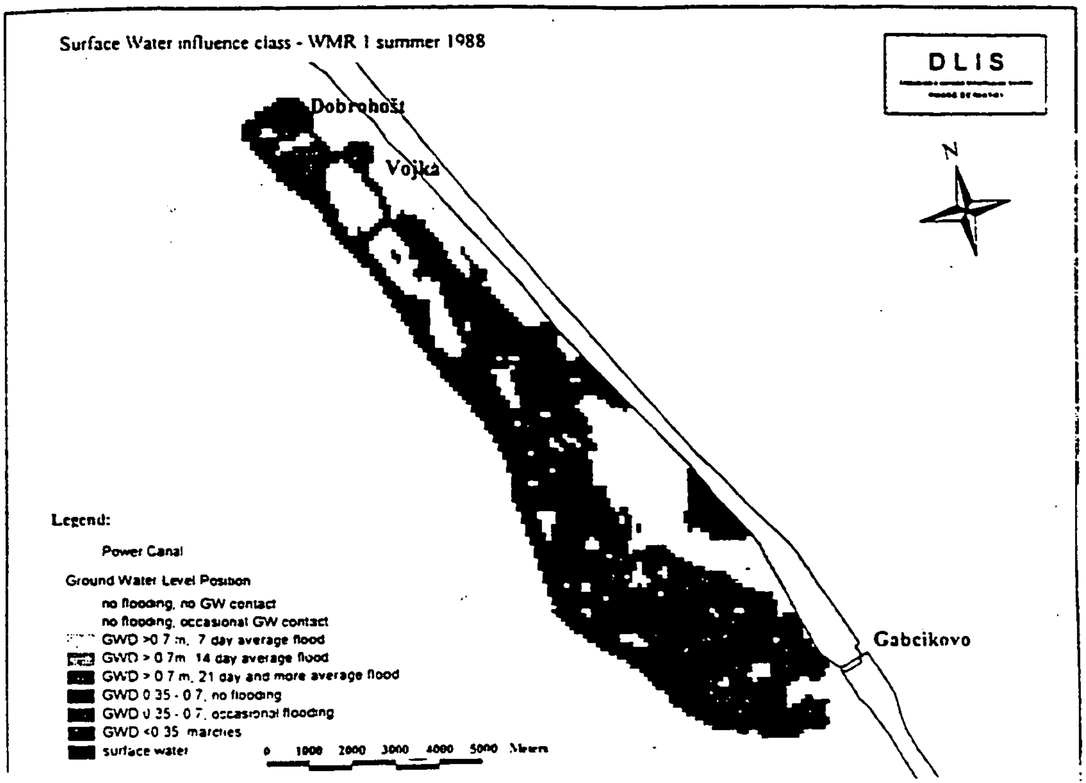


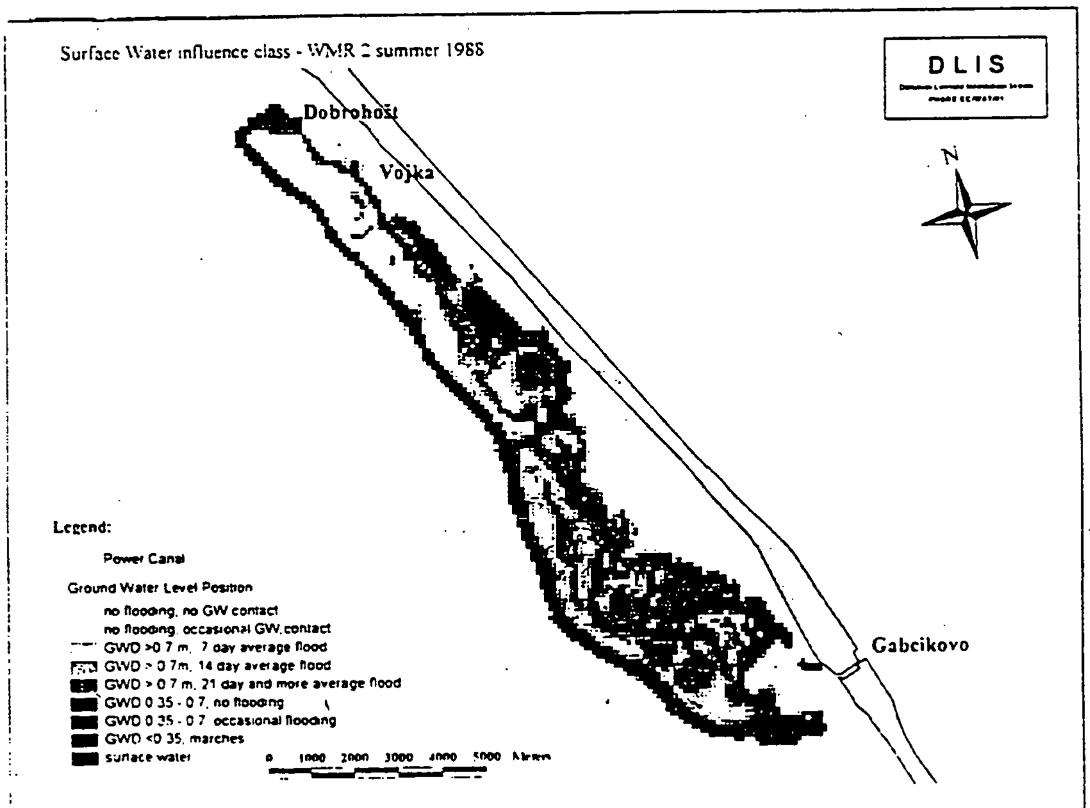


Figure 5. Hydrological regime in the river branch system categorized in ecological classes, corresponding to a situation with 1988 discharge data for pre-dam conditions (above) and for post-dam WMR II (below).





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## 5. CONCLUSIONS

The hydrological and ecological system of the Danubian Lowland is so complex with so many interactions between the surface and the subsurface water regimes and between physical, chemical and biological changes that a comprehensive mathematical modelling system of the distributed physically-based type is required in order to provide quantitative assessments of environmental impacts.

Such integrated modelling system coupled with a comprehensive data base/GIS system has been developed. The individual components of the modelling system represent state-of-the-art within their respective disciplines. The uniqueness is the full integration with full couplings within the same time step. The integrated system makes it possible at a quite detailed level to make quantitative predictions of the surface and ground water regime in the floodplain area, including e.g. ground water levels and dynamics, grounds water quality, crop yield and nitrogen leaching from agricultural land, sedimentation and erosion in rivers and reservoirs, surface water quality as well as frequency, magnitude and duration of inundations in floodplain areas. Such information constitutes a necessary support for a rational water resources management.

The various parts of the integrated modelling system were calibrated and validated on the basis of comprehensive data from the area.

A few selected model applications indicate that water quality problems in the Danube are not likely to occur for discharges less than 400 $m^3/s$, even in situations with underwater weirs. Furthermore, model results from the river branch system indicate that long term ecological changes are likely to occur in this area if the present operation continues.

## ACKNOWLEDGEMENT

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FORUM – FLOOD 1997, 10-12 October 1997, Krakow, Poland

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**Recent European Experience in River Flood Management**

## GENERAL

In the past catchment and river engineering has often been based on procedures and design methods, which were too narrow in focus. In many places river training and urbanisation have caused unforeseen negative consequences on groundwater, water quality and on flooding conditions. It is increasingly acknowledged that catchment and river engineering needs to be based on a more holistic technology and approach than traditionally. Many rivers in Europe are flowing across boundaries, and a harmonisation of methodologies is required between the countries.

In recent years severe flooding has occurred in several parts of Europe, both as localised flash floods and as basin-wide floods on major river systems. By their nature floods are generated by the coincidence of several meteorological factors but man's use of the river catchment also has an important impact upon the severity and consequences of the events. Although the investment in flood control structures has steadily increased during the second half of this century, the damage caused by floods has also increased dramatically.

After the severe floodings in the Rhine, the Rhone, the Po and other rivers in the early 1990's, the European Commission organised an expert workshop to discuss the current state-of-the-art and the needs for improvement in the area of river flood management. It was decided to launch a Concerted Action on River Basin Modelling, Management and Flood Mitigation: RIBAMOD, with the aim to

- identify difficulties arising from past management practices,
- identify the state-of-the-art in its area,
- identify best practise,
- take an overview of current EU research projects in the area, and
- identify research needs,
- provide scientific basis for decision makers.

The first expert meeting was held at DHI in Copenhagen in 1996, where a need for cross boundary development and collaboration in this field clearly emerged, that is

- risk assessment procedures and terminology,
- integrated modelling and decision support
- long term assessments of the environmental impact of structural and unstructural measures
- needs for common databases and standards

It is regarded essential, that new flood control and management concepts are properly evaluated on a regional basis by use of integrated models. These models should be used to evaluate the impacts not only on short term but also on longer term.

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The traditional approach to flood control is being reshaped these years to a flood management oriented concept. Such concept operates with different levels of flood protection for different areas - the highest level for the urban areas and the lowest for agricultural areas. The implementation includes flood forecasting and emergency planning tools, which facilitate effective management during the flood event.

**Skjern River, a sustainable river management solution.**

An example, where a sustainable river management solution is presently being implemented, is the Skjern River in Denmark. River training works were carried out on the lower 20 km of this river in the sixties turning 4,000 ha of meadows and marshland into farmland. The regulation had a severe effect on the water quality both in the river and the downstream fjord, with frequent occurrence of alga blooms in the fjord. But also from a flood control point of view, it did not work satisfactorily. In March 1972, soon after the completion of the original regulation project, a severe flood occurred in the Skjern River following a sudden spring thaw. It caused substantial damage in the river valley, when 50 mill. $m^3$ of water bypassed the river embankments breaching road embankments, destroying several bridges and flooding large areas of farmland.

A detailed model of the situation was established using the one-dimensional hydrodynamic modelling system MIKE 11 (including a GIS representation of the floodplains). By use of the model, it could be concluded that the new restored river would be able to accommodate the 1972 flood event without damage to infrastructure or unintended inundation. Under post-restoration conditions, a part of the river valley will contribute to the conveyance of large floods resulting in much smaller water surface slopes and velocities, thus reducing the risk of structural failure. Also from an environmental point of view, the impact will be highly favourable.

**Bangladesh, Flood Management Model**

Bangladesh is a flat delta at the confluence of three major river systems of the world: the Ganges, the Brahmaputra and the Meghna. With more than half of the country under the 12.5 m contour, about 30% of the cultivable area of Bangladesh is flooded in a normal year. An estimated 50% is vulnerable to either monsoon or tidal floods. Only 20% of the vulnerable area is protected. Flooding in Bangladesh is commonly caused by a combination of several factors such as overbank spilling of the main rivers, runoff generated by heavy local rainfall, and cyclone tidal bores or storm surges. The 1987 and 1988 floods were two of the most severe on record causing widespread damages to crops, urban areas and infrastructure and resulting in more than 3000 lost lives.

**Concepts of the Flood Management Model (FMM)**

Flood management is concerned with making decisions based on policies reflecting the needs of communities and the environment. It is complex and often without solutions which fully satisfy all concerned parties. The many components such as land use, environment, infrastructure, flood control structures, irrigation needs, agriculture, economics, society, fisheries, flood preparedness and flood forecasting render decision making and policy formulation extremely difficult.

Modelling floods in Bangladesh has been carried out by MIKE 11 flood models. The models provide outputs such as flood levels along the rivers and over the flood plains and, more importantly, simulate the impacts of interventions on flood levels. However, flood models do not produce the flood maps needed for identifying and prioritising flood management zones, nor do they produce maps of impacts on flood levels which greatly assist in assessing alternative solutions and carrying out multi-sectoral flood impact analyses.

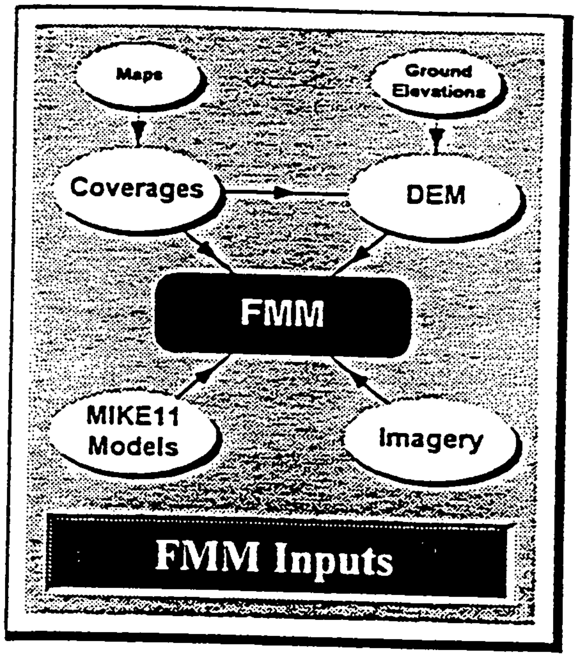
To produce these maps and to perform multi-sectoral impact analyses GIS technology is required. Flood depths and levels are represented as layers of data in the GIS which can be geographically related and analysed with data from other flood management components. The maps and results of multi-sectoral analyses are easily assimilated using a combination of graphic and statistical formats.

The Flood Management Model (FMM) is an integrated MIKE 11-GIS modelling system. FMM has the potential to assist in clarifying and disseminating information through enhanced mapping of impacts on flood levels, communities, agriculture, fisheries and the environment. The maps can also help provide project design specifications, monitor and assess the performance of flood control and drainage structures, and help distribute flood forecasts in a readily acceptable form to the general public.

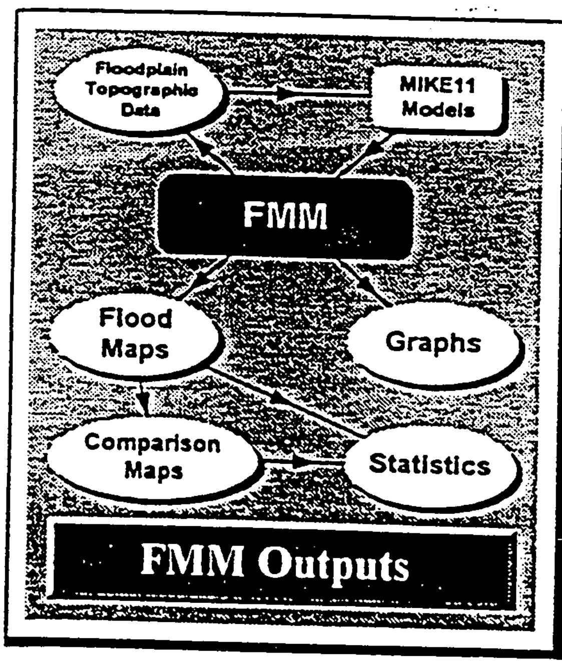
## FMM's Role in the Flood Management Cycle

Flood management follows a cyclic pattern linking ideas, proposals, consultations, adopting proposals, preparing guidelines, design, construction/implementation, and operating and maintaining finished schemes. This is the flood management planning, design, implementation and operation cycle, in which FMM plays a useful role. Fig. 1 shows the structure of a Flood Management Model.

At the planning level FMM helps assess proposed flood mitigation options and prepare environmental impact assessments. For design, FMM functions as a tool for determining civil works design criteria, designing structure operation rules, and providing inputs to flood preparedness programmes. At the implementation stage, FMM may be useful for a range of needs from scheduling flood prone construction works to a flood preparedness training aid. Real-time FMM operation linked with flood forecasting can help guide structure operators and assist emergency relief operations. FMM can also help present the consequences on flooding due to repair and maintenance of structures.




*FMM requires inputs of coverages (roads, rivers, settlements), a DEM, flood models and, if available, satellite and photo imagery.*




*FMM outputs topographic data for use in flood models and post-processes flood model simulation results into flood maps, comparison maps and graphs. Statistics are produced from the flood and comparison maps.*

Fig. 1. Inputs and outputs of the Flood Management Model

**Examples of FMM Outputs**

Flood Depth Maps

Flood depth maps show in graphical detail the variation in inundation depth over the floodplain, along with the flood-free areas. They give a clear picture of the depth and extent of flood inundation. The maps are produced using the results from a flood model simulation and can be at any instant in time or based on the maximum flood levels over the entire simulation. Fig. 2 illustrates a flood depth map.

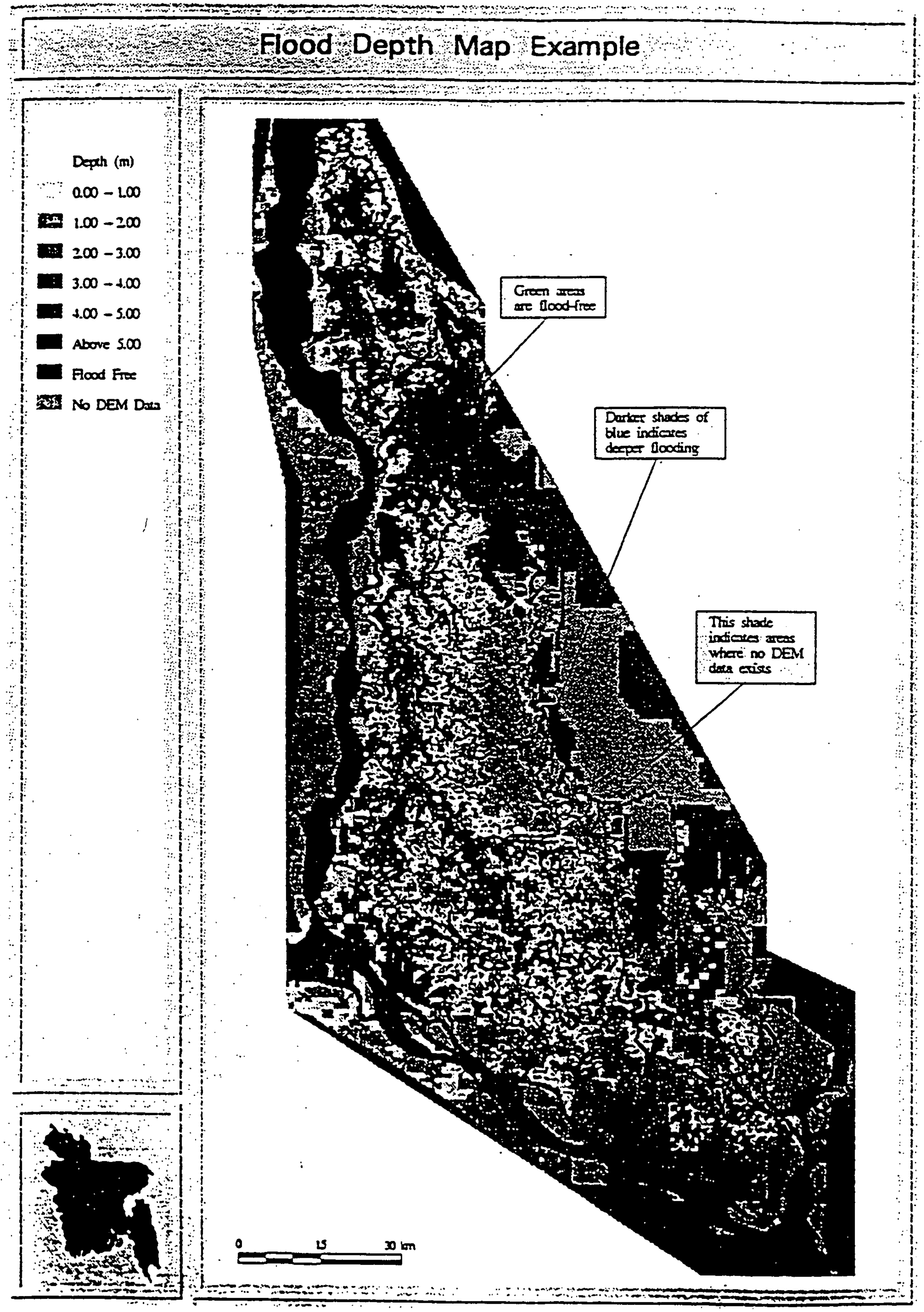



Fig. 2. Example of a flood depth map.

## Duration Depth and Crop Damage Maps

Duration depth maps are similar to flood depth maps, but take into account the critical duration of flooding (typically three days) which a crop can withstand without being damaged. From the duration depth map crop damage maps are produced. The normal procedure is to work with periods (10 to 15 days, or monthly) which correspond with the crop growth stages. The critical depths (the depth of water which will damage a crop if it is inundated for longer than the critical duration) must be supplied for each period. Fig. 3 shows an example of a duration depth map and a crop damage map.

## Conclusion

The recent floodings in Europe have called for a revaluation of the flood control concepts and procedures. The use of integrated modelling technology is essential to ensure a proper regional and long-term impact evaluation. The present tendency is developing towards integrated flood management solutions rather then pure flood control schemes. The models are indispensable tools for deriving such sustainable concepts, and for facilitating an effective dialogue between different interest groups along the river.

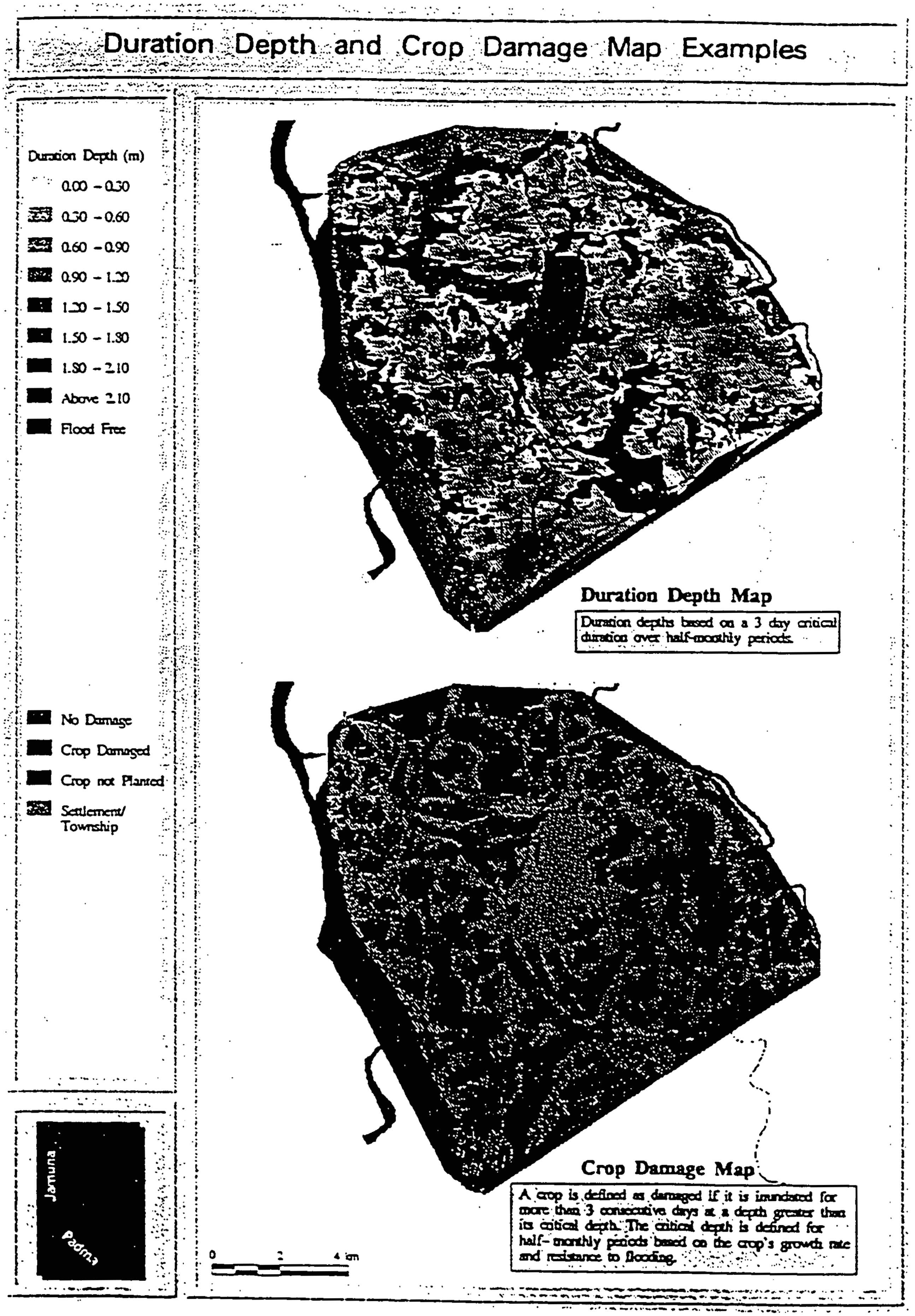



Fig. 3. Examples of duration depth and crop damage maps produced by the FMM.

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