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The use of hydro-ecological models in the Netherlands

Proceedings and Information No. 47
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CHO

The use of hydro-ecological models in the Netherlands

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Editors
J.C. Hooghart
C.W.S. Posthumus

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PREFACE

On 25 May 1993 the Technical Meeting 51 of the TNO Committee on Hydrological Research (CHO-TNO) was held in Ede, the Netherlands, on the theme: "The use of hydro-ecological models in the Netherlands". This meeting, organized in cooperation with the Dutch Association for Landscape Ecology (WLO), was a follow-up of the WLO-meeting "Hydro-ecological prediction methods for Policy and Management", held in Ede, the Netherlands, on 15 November 1991.

The main target of the meeting was to offer the potential users a better insight into the available hydro-ecological model types, and to show makers of models the possibilities of other methods.

This publication contains the papers presented at this meeting.

In the Netherlands one of the largest environmental stresses to vegetations in nature reserves is the **structural lowering of the ground water levels** and the physical and chemical alterations, triggered by this lowering, at the site of these vegetations. Man-made interventions in the water management cause this environmental stress leading to adversal ecological effects. In the Netherlands this environmental stress is called **"verdroging"**. In the English language an equivalent term does not exist, so in this publication different (Dutch) authors use different (litteral) terms. You will find the terms **"drought damage"** in nature conservation areas (Claessen, Witte et al.), **"desiccation"** (Latour, Barendregt et al.) and **"verdroging"** (Garritsen). Another term used in this publication is **"dehydration of wetlands and forests"**. In fact all these terms mean the same, but the taste of the authors is different.

Delft, December 1993

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FUTURE QUESTIONS OF POLICY AND MANAGEMENT FOR APPLICATION OF HYDRO-ECOLOGICAL MODELS

F.A.M. Claessen

ABSTRACT

The questions for future application of hydro-ecological models are investigated by means of:

i) a small inquiry among ten institutions involved with the future policy and management of water dependent ecosystems in the Netherlands;
ii) an analysis of the topics presented in the recently issued National Research Programme on Dehydration of Wetlands and Forests (Nationaal Onderzoekprogramma Verdroging - NOV).

An evaluation is made to what extent the available hydro-ecological models meet the demands of institutions involved in water management. For this evaluation a comprehensive review of six operational models by Wassen and Schot (1992) is used.

Concerning the model users, a distinction can be made between questions for model application from local authorities and the questions from provincial and national authorities.

Concerning the models a distinction can be made between two general types of hydro-ecological models: the "top down" and the "bottom up" approaches. The model ICHORS (= Influences of Chemical and Hydrological factors On the Response of Species) represents the bottom up type, DEMNAT (= Dose Effect Model for Terrestrial NATure) the top down type. MOVE (= Multiple stress On VEgetation) represents a mixture of aspects of both types.

The major goal of the scientific meeting is improvement of the communication amongst the makers of different models, as well as between the makers and the users of models. When comparing the demands for application of models via the small inquiry and those of the NOV, it seems that most demands coincide.

1 INTRODUCTION

During the last decades much work has been done in the Netherlands to develop methods for quantifying the consequences of interventions in management of surface water and groundwater on ecosystems, both aquatic and terrestrial. The models that have been made contain explicitly written decision rules about intervention-impact relationships. Partly these rules are founded on expert judgement, partly on statistical analysis of field data, and partly they have been derived from generalized knowledge of field data (typology or classifications of ecological plant groups or soil types). Because the models were developed at different institutions, although for simular goals, there are now several, hydro-ecological prediction models in the Netherlands that have been developed independently of each other. Each model is connected to its own specific network of research institutions and users. To mention a few examples: the model ICHORS, developed at the University of Utrecht in cooperation with the administrations of the provinces of Noord-Holland, Zuid-Holland and Utrecht; DEMNAT-2, developed by the Institute for Inland Water Management and Waste Water Treatment (RIZA) and the National Institute of Public Health and Environmental Protection (RIVM) in cooperation with the University of Leiden and the Wageningen Agricultural University; NTM (Technical Nature Model), developed by the Institute for Forestry and Nature Research in Leersum in cooperation with the Winand Staring Centre for integrated Land, Soil and Water Research in Wageningen.

At the meeting "Hydro-ecological prediction methods for Policy and Management" of the Dutch Association for Landscape Ecology (WLO), held in Ede, the Netherlands, in 1991, Wassen and others presented a clear review of the state of the art of different hydro-ecological methods in the Netherlands (Wassen and Schot, 1992). Six models were compared and evaluated with respect to several criteria. Wassen concluded that many methods or models deal with the same kind of questions. But the used approaches differ considerably as do the scales of the models (from local to national). Although, during this meeting not everyone agreed with him, this writer felt the existence of different paradigmas. Paradigmas that express themselves in different ways of thinking, different concepts, developing in an isolated environment of research and application, sheltered from other "breeding beds". Not only scientific but also socio-cultural aspects determine these differences. In Ede was concluded that the communication between the builders of the different methods should be improved. Also, that potential users should become well informed about the specific qualities and values of the models for specific management and policy problems.

For these reasons, a new exchange of information between hydro-ecological model builders and users was organized on 25 May 1993 in Ede, the Netherlands. This meeting of the TNO Committee on Hydrological Research (CHO-TNO), organized in cooperation with the WLO, had to improve this communication. The main target of the meeting was to offer the potential users a better insight into the available models and model types, and to show makers of models the possibilities of other methods.

A small inquiry was undertaken under potential users about the demands for application of the models in the coming five years. For the same reason the NOV was examined and compared with the inquiry results.

2 PROGRAMME OF THE MEETING

The programme of the meeting was made with the following considerations in mind:

. The meeting is a logical continuation and not a repetition of the WLO-meeting in Ede 1991. The participants are recruited from the population of both model makers and users. The users are questioned about their specific needs in the coming years concerning the application of the methods and models for policy-making and environmental management;
. Three different model types will be presented: ICHORS by Barendregt and Nieuwenhuis (1993, this volume), DEMNAT by Witte et al (1993, this volume) and MOVE by Latour et al (1993, this volume);
. An independent hydro-ecologist (Garritsen, 1993, this volume) will evaluate these presentations with respect to the needs of the model users;
. Finally, a panel of hydro-ecologists (Nieuwenhuis, Roelofs, Kemmers, Runhaar, Jansen, Pedroli, chaired by Claessen) will discuss several questions posed by Garritsen and the audience. Central questions will be:
  - What kind of policy and management problems have to be tackled during the next five years?
  - Which are the basic principles of the presented methods? Is a combination of methods promising? What are the similarities in the used methods? What is the in- and output? How does the response function look? What is the scale of the model? In what kind of areas can the models be applied?
  - Is it desirable and feasible to develop better models? Which points of model research are to be favoured?

3 FUTURE QUESTIONS FOR POLICY AND MANAGEMENT

A small inventory of the future needs of model applications was carried out, concerning the ecological impact of water management and environmental interventions. The following institutions were questioned:

- 4 national institutions;
- 2 provincial policy boards;
- 1 nature conservation organization;
- 2 regional waterboards.

Six questions were posed. The questions and the main answers will be described.

1. Type of nature area?

   Most institutions agree that the important study areas are the wet, moist and riparian sites in nature conservation areas or farmland with elements of high ecological values. Figure 1 presents the area in the Netherlands where hydro-ecological modelling is most needed. The black and grey coloured areas are nature conservation areas effected by drought damage. This damage has occurred since 1950. The black areas have the highest priority for restoration (Claessen, 1993).

2. Kind of interventions?

   Answers were: changes in drainage facilities, changes in the inlet of water; changes in the extraction of groundwater; changes in the emission of compounds that threaten the quality of the environment; increase of the evapotranspiration of crops; restoration measures; climatic changes; changes in land use.

3. Scale?

   For local management a scale of 1 : 5 000 to 1 : 25 000 will suffice, for policy and management purposes at national or provincial model scales of 1 : 100 000 to 1 : 500 000 are useful. For still larger areas a scale of 1 : 10 000 000 will suffice.

4. Target variables?

   For most institutions these variables are plant species or groups of plants (vegetations). But insight in the impact of interventions on birds and other animals as target variables is also wanted.

5. Policy targets?

   The local authorities and managers want to get answers to questions related with nature conservation, recovery and development. The national institutions especially are concerned with the environmental quality and formulating general emission standards for toxic and non-toxic compounds.

6. Experience with hydro-ecological models?

   Some local and provincial authorities have experience with ICHORS, ECAM and HYVEG; the national authorities especially have experience with NTM and DEMNAT.

The answers are not surprising. However, it was possible to make a distinction between the wishes and demands of the local and regional users of models and the users at provincial and national level. The first group wants detailed studies focussed on a few sites; the last group wants more global studies for large areas. The first group is mainly interested in the impact of water management interventions, both inside and outside nature areas; the last group is especially interested in the impact of measures outside nature areas. Climatic changes and sea-level changes are issues of interest only for national policy. Another topic of interest is the specific contribution of the different components to the total environmental stress (major multi-stressors: eutrophication, acidification and drought damage).

Up to now, some local and provincial users prefer ICHORS, the national institutions use DEMNAT and the ecotope classification system. Nature conservation organizations have no special preference.

Nature conservation areas affected by drought damage to wet and moist ecosystems
Source: RIZA / RIVM / Provinces

Area affected by drought damage

Area affected by drought damage with priority for restoration

Figure 1 The black and grey spots represent the total area affected by drought damage
The black areas have priority for restoration.

3.1 The National Research Programme on Dehydration of Wetlands and Forests

Drought damage on vegetation and fauna is the negative effect of a structural drop of the groundwater level. This effect can also be a result of a decrease in seepage intensity or the inlet of polluted water to compensate for the groundwater shortages.

The National Research Programme on Dehydration of Wetlands and Forests (Nationaal Onderzoekprogramma Verdroging - NOV) is a proposal for research in the period 1993-1996. The programme was made by different policy and research institutions, operating at national and regional levels. The short-term target of this programme is to put into use the expertise and tools for execution of the drought-damage combatment policy. The long-term target is to set up and stimulate research, which leads to the improvement of these tools.

The following topics in the programme are relevant with respect to the future need for application of hydro-ecological models:

- There is a need for further development of tools to quantify and monitor drought damage in nature areas. It is preferable that these tools for monitoring drought damage and the target variables of eco-hydrological models should be compatible;
- A second need is to specify target and reference situations for drought damage areas, to clarify the goals and aims for recovery and development of nature conservation areas. Regarding this need the same remark can be made on the compatibility with hydro-ecological prediction models. If they are compatible, it will be possible to compare the monitoring results with the results of model calculations in future;
- A third need in the programme is to develop a method to define these target situations in terms of abiotic conditions. Hydro-ecological knowledge and methods are a prerequisite for this step;
- A fourth topic of research is an analysis of the contribution of eutrophication, acidification and drought damage to the distribution and quality of aquatic and terrestrial ecosystems;
- Another topic is the quantification of the relation between hydrology, pedology (soil) and vegetation. Also of importance is the impact of inlet of surface water of poor or chemically anomalous quality on aquatic ecosystems;
- Finally, there is a need for the development of tools for cost-benefit analysis of mitigating and compensating measures by regional and provincial authorities. Regarding this need, decision support systems are mentioned as a useful product.

Comparing the demands from the small inventory with the NOV leads to the conclusion that a number of topics coincide. New elements in the programme are especially the definition of the target and the reference situation and the need for cost-benefit analysis and decision support systems.

4 AVAILABILITY OF MODELS AND METHODS

At the WLO-meeting in November 1991 a comparison was made between six models, which were operational at that time. The goal was to illuminate the use of these models for specific questions to the potential users (Wassen and Schot, 1992).

The following models were compared: ICHORS, HYVEG, ECAM, DEMNAT, NTM and the Ecotope classification system. The last two tools are not real computer models but simple pragmatic decision-supporting tools. The models were compared on a number of aspects: type of intervention, theoretical principles and concepts, model input and output, calibration and validation, actual applications.

The most important conclusions of the comparison are:
- most models deal with water quantity and to a small extent with water quality (except for ICHORS);
- emphasis is on terrestrial ecosystems and hardly on aquatic ecosystems (again except for ICHORS);
- some models have a top down approach, starting from a typology or classification based on generalized hydro-ecological information. Presupposed is a causal-deterministic interrelationship between the occurrence of plant species and a limited set of site factors. The scale is mostly small, a small number of input variables is used. The other type of models has a bottom-up approach, starting with collected field data of plant species and site conditions. Presupposed in this case is that there exists a causal relation between plants and measured site conditions, if there is a statistical correlationship. Characteristics of this type of models are: statistical-correlation, many input variables, large scale, more specific than general;
- some models deal with plant species, other with plant groups or vegetation types. As yet no attention is given to birds and animals;
- the models are not yet well-calibrated and validated. The application of the models was often restricted to the study area for which, in the first instance, the model was developed.

Finally, from this comparison it appeared that the methods deal with the same kind of questions. Only the approach of a problem (for example, what kind of variables are involved), the scale of the model and the study goals differed to some extent.

In this book three models will be presented. Two of these models are existing models that have already been used: DEMNAT (of the top-down type) and ICHORS (of the bottom-up type). The third model MOVE (Multistress mOdel for VEgetation) is a mixture of the two types. This model is still in development and has some familiarity with ICHORS (the correlation approach with probabilistic output), but also with DEMNAT and NTM (Ellenberg indication figures, deterministic site module). Because of these differences and partial similarity it is worthwhile to present this model, too.

5 CONCLUSIONS

1) From the small inquiry and the former comparison by Wassen and Schot (1992) it follows that a distinction is to be made between the demands of local and regional institutions concerned with local management and the demands of the provincial and national institutions concerned with regional and national planning. About the kind of nature areas and the kind of the interventions and target variables involved, there is large consensus of opinion.

2) By comparing the inquiry results with the NOV it appeared that most demands coincide. New items in the NOV are the wish to specify target- and reference

situations, the wish to improve the empirical foundation of the response functions, especially in case of recovery of nature conservation areas and the need for cost-benefit analysis tools and decision support systems for regional and local authorities.

3) Finally, the most important questions that have to be answered by the makers of the three models are:
   - What are the principles of the method?
   - What are the similarities with the other methods? How far are these methods complementary?
   - What is the input, what is the model output?
   - How do the response functions work; what scale level is used?
   - What are the possibilities for applications with respect to area types and type of interventions?
   - Are there reasons for further improvement/refinement of the model?
   - What kind of research topics or aspects must be emphasized?

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ICHORS, HYDRO-ECOLOGICAL RELATIONS BY MULTI-DIMENSIONAL MODELLING OF OBSERVATIONS

A. Barendregt and J.W. Nieuwenhuis

1 INTRODUCTION

With the introduction of computers the modelling in hydrology, soil science and biology became quite common. Combining different models facilitated the modelling of whole ecosystems. Especially in eutrophication abatement many models have been developed to simulate the relation between nutrient concentrations and algal blooms (Van Liere and Gulati, 1992; Jørgensen, 1988). Modelling of freshwater marshes started fairly recently and the efforts are just beginning (Mitsch et al., 1988). Complex wetland modelling (e.g. Hopkinson et al., 1988; Mitsch and Reeder, 1991) needs information about elements of hydrology, primary production, retention and resuspension of nutrients and energy flow. However, hydrology mostly covers precipitation, evaporation-transpiration and surface water modelling. Only recently the actual role of groundwater in wetlands is recognized (Lehr, 1991; Richardson et al., 1992). Differences in groundwater flow result in differences in water chemistry and therefore in the availability of nutrients.

Concentrating on the typical Dutch conditions, the hydrology of wetlands is complex since most water is managed. Water balances and water flows of both groundwater and surface water are needed to describe the detailed hydrology in a region (e.g. Witmer, 1989; Schot and Molenaar, 1992; Van Bakel, 1992). This information is important for two reasons. Firstly, in the terrestrial systems the soil chemistry and the nutrient availability change by limited differences in phreatic level and as a result the primary production and the vegetation types change (Grootjans et al., 1985, 1986; Verhoeven et al., 1988). The desiccation of many Dutch areas is a problem for the ecological diversity (Fiselier et al., 1992). Secondly, the water balances are needed to describe the chemistry of the water in the ecosystems, since the proportion of the three main sources (precipitation, fresh groundwater, polluted river water) results in a different ecosystem development (Bloemendaal and Roelofs, 1988; Van Wirdum, 1991a, 1991b).

Computer modelling of ecosystems is performed with two objectives. Many relations in an ecosystem are not fully known and generalisation from field data and calibration is needed. On the other hand, if these relations are known, the computer can easily calculate very complex relations and a number of different options can be compared.

These models give the effects of a special human change in the ecosystems, in this case a change in hydrology. From the point of view of the management, these models should be applicable in most ecosystems with effects on the ecosystem and the characteristic species. From the point of view of science, these models help to understand how ecosystems function. However, the higher the ecosystem level, the higher the generalization. The characteristic differences within bogs, fens, meadows, rivers, rivulets, ditches, shallow lakes, etc., cannot be taken into account when all these ecosystems are incorporated in one model. Moreover, if types of vegetation are to be predicted, the regional characteristics are needed, for instance to predict *Carex aquatilis* to be present only in the northern parts of the Netherlands and *Carex pendula* only in the most southern parts.

The construction of a model needs a selection of variables and dimensions in application. Four steps are needed in order to inform about the effects of changes in hydrology in the biotic composition of an ecosystem. First, the hydrological changes by human impact will initiate differences in the flow of water. The second step is that these changes in hydrology will result in different processes in the landscape, such as nutrient availability, salinity, aeration and transport. These changes might be local or regional: they result in different non-biotic conditions in the habitat of the species. The third step is the direct relation between the conditions in the habitat and the ecological preference of the species. The last step is the evaluation of the results of the former steps, where the different hydrological options are compared with the target condition of the policy. The second step especially is complicated since the changes in the processes are complex, non-linear and full of feed-back mechanisms; the third step needs information on the direct relations of species to a complex of physical and chemical variables. Unless these relations can be obtained from literature, new information from field data has to be added.

2 MODELLING OF AQUATIC ECOSYSTEMS IN POLDERS

The management of the ditches and small lakes in the polders in the western parts of the Netherlands needed a tool to evaluate possible changes in hydrology. The relation between the presence of species and a great number of habitat conditions was requested for these aquatic ecosystems. The ICHORS-model was constructed to predict the Influence of Chemical and Hydrological factors On the Response of Species (Barendregt et al., 1986, 1993). Hydrological models could support ICHORS and the evaluation of the results was separated from the predicting model; the most urgent information was needed for the modelling of the habitat.

The first question in the modelling was the definition of the biotic variable. Policy and management wish to be informed about the conditions of an ecosystem. However, an ecosystem is an abstract definition, not to be recognized in the field. A type of vegetation is also a synthesis, but it is possible to point out in the field, with the benefit that the vegetation forms the structure in the ecosystem where the animals react on. In practice it is still a problem to define where a gradient ends the type and another type starts. Moreover, if the complete type contains a maximum of 20 species, and only 6 are present, is it possible to label this vegetation? Or do we have to confine to the very characteristic species, which are always rare, due to their narrow ecological amplitude? Based on these problems we have chosen the level of the plant species. Without difficulties the presence or absence of a species can be established in the field. One restriction has to be given: only plant species that depend on water in the rooting zone have to be incorporated. For that reason only the hydrophytes and phreatophytes according to Londo (1988) are modelled.

The next question is which variables influence the presence of the hydrophytes and phreatophytes directly in the habitat. From literature a number of variables are known:
- physical dimensions of the aquatic system (width, depth);
- the physical morphology of the shore line;
- type of soil at the shore line and in the water bedding;
- amount of sediments with their chemistry;
- water levels and their fluctuations;
- water chemistry with characteristics of salinity, nutrients, acidity, oxygen, alkalinity, etc. with their fluctuations;
- turbidity of the water;
- groundwater flow with influences on carbon-oxides;
- direct human influences, such as application of pesticides, management and shipping.

These are the variables to be modelled in the habitat-model. The processes which change the habitat-variables are often connected with hydrology, such as artificial drainage or flooding, pumping of surface water that changes the ratio between the sources of water, abstraction of groundwater that changes flows, etc. The chemical reactions, such as precipitation of phosphorus or mineralisation of organic soils, have to be incorporated. Moreover, the vegetation starts succession with accumulation of organic matter and nutrients.

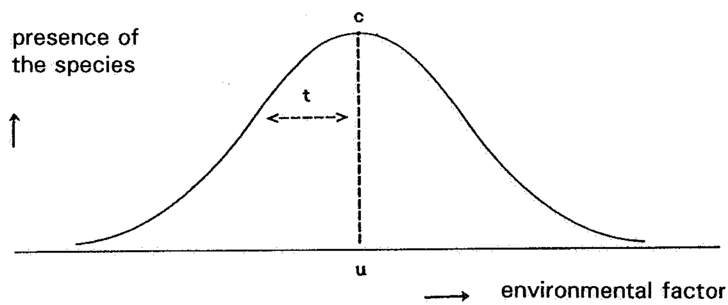
3 DATA COLLECTION

The most easy way to model specific relations is to obtain the data from literature. In our case this was not possible since there were no publications with all the required values of the variables and the scarce data available were not appropriate to be used. The second way to be informed is to use different existing raw datasets. This combination of datasets was not possible, since the data have been collected at different locations or at different times and only a restricted number of variables were included. The only other alternative was to collect new data in the field. This takes time and energy but has the benefit that the data were collected in a standardized way: all variables at the same time with the same methods, so that the data are multi-dimensionally comparable and include biotic and non-biotic variables.

In practice a number of decisions have to be taken. In the optimal condition one samples a habitat with all its variables and dimensions in space and time. Technically this cannot be realized and restrictions were made. First, the fluctuations during the seasons were omitted by restricting the sampling to the summer period. This is the season when the plant species are present and will be influenced most by the habitat conditions. To omit locations with instable conditions, only locations without hydrological change during the last years were collected. The next step is the definition of the habitat to be sampled. In the field we used homogeneous areas of 0.5 to 5 ha. in which no variation was to be detected in soil, hydrology, type of vegetation, water chemistry and physical variables. A wide range of variables was estimated; only sediments, oxygen and management are important variables which were not collected for technical reasons. The aim was to collect as many locations as possible with a wide geographical distribution, to obtain a very diverse dataset. Special attention was given to the multi-dimensional distribution of the data: all possible combinations of independent variables were collected in a range as wide as possible. The resulting dataset with hundreds of observations describes all possible habitats in the field, from fresh to saline, from oligotrophic to eutrophic, from acid to neutral, from clay soils to peat or sand, etc.

4 MODELLING TECHNIQUE

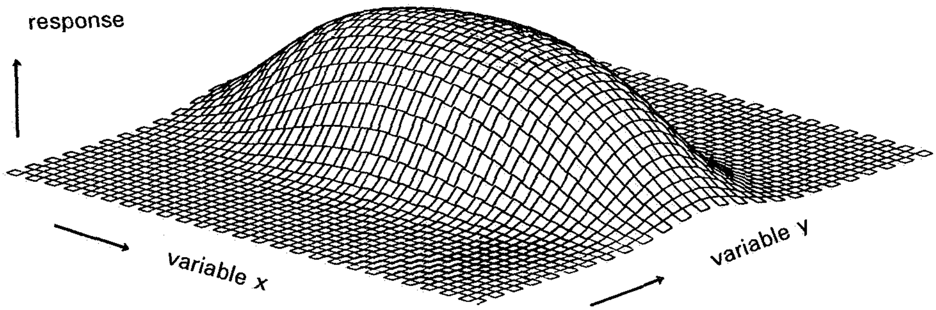
If the dataset is an accurate description of the total variation of a large area, the variation can be used to generalize the relation between environment and plant species. At that moment some restrictions and rules are valid in constructing a model that predicts the presence of each plant species. The ecological principle that each species has its characteristic optimum curve (Austin, 1976; Olff, 1992; Huisman et al., 1993) and at the same time the relation with a number of variables is important, needs a specific calculation method which uses the explaining variables only. Moreover, the optimum and tolerance of the optimum-curve is important and both continuous variables (such as phosphorus) and discrete variables (such as soil type) have to be integrated in one method which is easy to apply in practice.



$$p = e^{f(x)} / (1 + e^{f(x)}) \text{ , where } f(x) = ax^2 + bx + c$$

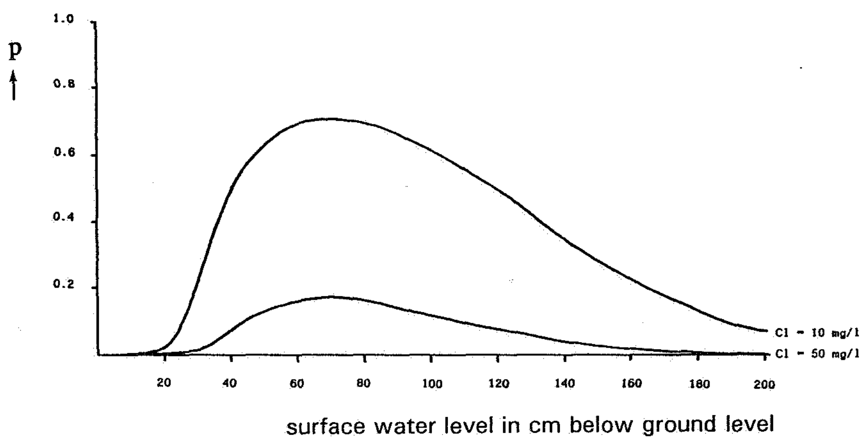
Figure 1 Optimum curve of a species with optimum (u), maximum (c) and tolerance (t), with the general regression formula.

These options lead to the use of logistic regression methods (Hosmer and Lemeslow, 1989) from the group of Generalized Linear Models (Nelder and Wetherburn, 1974). For each species and for each variable the optimum curve was calculated (Figure 1), with its optimum (u) and tolerance (t) (see: Ter Braak and Looman, 1987). To obtain a binomial distribution the values of the response variable are transformed to presence-absence data. The probability of the presence of the species can be calculated with the formula in Figure 1. To handle the problem that not all the optimum curves will be explaining in equal scale and that variable might be correlated, a multiple logistic regression method was chosen with a foreward selection procedure. This procedure in GENSTAT (Alvey et al., 1977) adds in each step the linear and quadratic terms of each continuous variable and the terms of each discrete variable. The variable that decreased the analysis of deviance (Nelder and Wetherburn, 1974) most, was selected at each step and added to the model. This procedure was stopped when the analysis of deviance decreased below 3% of the starting deviance or when seven variables were added to the model, to avoid overfitting. This last remark is also valid at each calculated species since we used the minimum number of ca. 25 positive observations in the dataset.



$$f(x,y) = a_1x^2 + b_1x + a_2y^2 + b_2y + c$$

Figure 2 Two combined optimum curves of a species with the general regression formula.



$$f(x) = -0.08*(Cl^{-}conc)^2 - 3.25*Cl^{-}conc - 15.98*(level)^2 + 58.92*level - 50.11$$

Figure 3 Example for a multiple regression formula of *Achillea ptarmica*: optimum curve of surface water level below ground level with log 10 chloride concentration of 10 en 50 mg/l.

In Figure 2 the optimum curves of variable x with a wide tolerance and variable y with a narrower tolerance are combined in a multiple optimum curve. It can be indicated that the probability of presence of the species will only be high if the values of both variables are optimal. This is illustrated with the example of the relation between the values of two variables influencing the presence of *Achillea ptarmica* (Figure 3). The continuous curve of the level of surface water below ground level is quite different at a chloride concentration of 10 mg/l and the curve at 50 mg/l. Both optima are at 65 cm below ground level, but the response decreases from 0.70 to 0.18.

The final regression models were incorporated in the ICHORS computer program written in FORTRAN. All models resulting from a dataset are installed in a version of this program. If values are attached to the incorporated variables, the program will result in the requested probabilities of all species.

5 MODELLING RESULTS

Through the years six versions of the ICHORS-model have been constructed:
- Version 1.0, based on a dataset (n=773) with surface water observations in the Vecht River plain and in saline polders near the village of Groet, containing 145 regression formulas (Barendregt et al., 1985);
- Version 2.0, based on a dataset (n=306) with observations of phreatic water in marshlands in the province of Noord-Holland, containing 122 regression formulas (Barendregt and Wassen, 1989);
- Version 3.0, based on a dataset (n=770) with surface water observations in the province of Noord-Holland, containing 163 regression formulas (Barendregt and Wassen, 1989);
- Version 3.1, based on a dataset (n=193) with surface water observations in the lakes and peat ponds in the Vecht River plain, containing 76 regression formulas (Barendregt and Bootsma, 1991);
- Version 3.2, based on a dataset (n=745) with surface water observations in the western parts of the province of Utrecht, containing 135 regression formulas (Barendregt and Bootsma, 1991);
- Version 3.3, based on a dataset (n=645) with surface water observations in the eastern parts of the province of Zuid-Holland, containing 174 regression formulas (Barendregt and Van Leerdam, 1993).

In the four ICHORS-3 versions 4.5 variables (average value) are incorporated in the regression formulas. No dominating variable is present in any formulas to explain the presence of the species (Barendregt, 1993) nor in groups of variables (Table 1). Another indication can be obtained from the first selected variable in the stepwise calculation of the 182 regression equations of ICHORS 3.3. Six variables were selected in more than 5% of the cases. Three variables describe the physical environment (level of surface water, soil of the water bedding and width of the surface water, with 15%, 15% and 6% respectively); three variables describe the chemical environment (phosphate, magnesium and iron-concentrations, with 13%, 9% and 7% respectively). These chemical variables can be explained by the influence of nutrient levels (Bloemendaal and Roelofs, 1988; Moss, 1988; Kooyman, 1993), salinity (De Lange, 1972; Haller et al., 1974; Van den Brink and van der Velde, 1993) and iron toxicity (Snowdon and Wheeler, 1993).

Table 1 Relative importance of groups of variables expressed as a percentage of the variables incorporated in all regression equations in the four ICHORS versions. Number of variables per group indicated in brackets.

ICHORS-version	3.0	3.1	3.2	3.3
number of regression equations	163	76	135	182
average number of variables used	4.8	4.0	3.8	5.4
(4) water level, width, depth, sapropel	22	29	14	21
(1) turbidity	3	3	2	4
(2) soil of water and bank	18	4	18	11
(2) seepage/infiltration (summer/winter)	4	8	8	2
(1) pH	4	6	5	6
(3) anions (HCO_3, Cl, SO_4)	8	9	7	8
(3) cations (Na, Mg, Ca)	14	7	12	12
(5) nutrients (P, N, K)	14	18	18	18
(2) Si + Fe	4	8	8	12
(4) water typology (ion index)	8	7	8	6

Table 2 Tests of six species models with independent data. From the dataset (n=770) 391 random observations were used to calculate the regression-formula. With the other 379 observations the formula was tested with input of the non-biotic data. The resulting 379 predictions were divided into 4 (5) classes with increasing values (number of observations indicated at the species) and compared with the percentage of actual presence of the species in the classes.

Butomus umbellatus	predicted		2.1	17.1	30.7	73.8
(n=208-82-30-59)	observed		1.9	15.9	23.3	54.2
Caltha palustris	predicted		2.8	15.7	31.1	62.6
(n=241-80-31-27)	observed		4.9	11.3	29.0	29.6
Elodea nuttallii	predicted		3.2	17.2	32.7	60.4
(n=157-73-39-110)	observed		4.5	21.9	28.2	64.5
Equisetum fluviatile	predicted	0.9	5.8	17.0	39.1	74.2
(n=86-69-84-75-77)	observed	0.0	10.1	27.4	33.3	76.9
Glyceria maxima	predicted		8.4	30.0	50.0	82.6
(n=101-45-36-197)	observed		16.8	44.4	58.3	83.5
Potamogeton natans	predicted		3.1	15.7	29.8	69.1
(n=267-55-22-35)	observed		5.6	16.4	13.6	45.7

Since there are no independent variables that can be improved with new information, it is not possible to calibrate the regression models. A validation is possible, if there is an independent dataset available with both non-biotic and biotic data from the same region. Since the collecting of such a dataset takes a lot of time, tests can be hardly executed. A solution to this problem is to divide the total dataset randomly into two parts. With the first part regression formulas were calculated; these formulas will be less reliable than those from the total dataset. It is possible to test these restricted calculations with the second part of the dataset, since the non-biotic variables of each observation are the input to the model and the resulting probability can be compared with the actual presence of the species. With a division of the predicted values in classes, the percentage presence of the species can be indicated. Table 2 summarizes the results of six species models. The conclusion is that these species are predicted in a correct way.

Table 3 Tests of ICHORS with four independent datasets. The non-biotic proportions in these datasets are the input of ICHORS. The obtained predicted values of the plant species are divided into eight (seven) classes with increasing values. For these classes the mean observed presence of the plant species in the datasets is calculated.

class number	1	2	3	4	5	6	7	8
interpolation in Noord-Holland of ICHORS version 3.0 at 50 locations (8150 predictions):								
n=	1917	1365	1245	812	839	779	705	488
mean predicted response	0.0	0.4	2.7	7.4	14.5	27.0	46.3	76.3
mean observed presence	4.2	11.9	20.9	33.9	41.7	56.6	66.3	71.7
extrapolation in Eemvallei (data June 1988) of ICHORS version 3.0 at 62 locations (10106 predictions):								
n=	2052	1409	1363	868	1182	1070	1163	999
mean predicted response	0.0	0.4	2.7	7.3	14.6	26.8	46.7	78.2
mean observed presence	0.9	3.6	4.6	11.6	16.5	24.2	36.6	57.1
extrapolation in Eemvallei (data August 1988) of ICHORS version 3.0 at 62 locations (10106 predictions):								
n=	2071	1341	1417	891	1132	1123	1149	982
mean predicted response	0.0	0.4	2.7	7.4	14.6	27.0	46.5	78.2
mean observed presence	0.9	4.3	4.1	11.1	15.6	25.8	35.9	58.8
extrapolation on the shore of IJsselmeer of ICHORS version 3.2 at 95 locations (12825 predictions):								
n=	1261	1587	2108	1520	2648	2166	1535	
mean predicted response	0.0	0.5	2.7	7.2	17.1	36.5	64.8	
mean observed presence	1.6	7.6	10.8	11.8	20.9	30.6	44.9	

Another solution to test the ICHORS model is the application of another independent dataset not collected for this purpose. For the province of Noord-Holland it was possible to construct a dataset from archives data to test the version 3.0 (interpolation). With three datasets from different regions (extrapolation, although with a comparable variation), versions 3.0 and 3.2 were tested for their predictions. The probabilities of all species were integrated to obtain a general view of the total model and to obtain a large number of data. The resulting data of the eight classes are presented in Table 3. Sometimes the conclusion is drawn that the predicted values fit well with the data, even in cases of extrapolation. The difference in the interpolation of the Noord-Holland dataset can be explained by the data collecting, since the provincial data per km^2 are used in the calculations instead of the ICHORS observations of ca. one ha. This will result in higher values of observed data.

The final test of the reliability of the model will be possible in future. The values of the incorporated variables and the presence of the species will have to be collected before and after a change in the hydrology or the environment. Since the model predicts the presence of species at stable conditions, a change in the hydrology or the environment will last for many years and only after this time the model can be tested. The complete sets with data from such a long period are not yet available.

6 APPLICATION OF ICHORS

The running of the program ICHORS needs four points of attention. Firstly, the method and for that reason the program is constructed to assess defined changes in aquatic ecosystems. This implies that the method is not generally valid in all cases. Secondly, the method needs (measured or predicted) values of many variables in the defined environment, and not only a global change in a special area. These values describe the non-biotic conditions in the environment in a scenario and are used as an input of the program. This step in the procedure needs most energy. Thirdly, the reponse calculation of the program is very easy. Within a computer-second the probabilities of encountering a large number of species are calculated. Fourthly, the results of the program are only these probabilities. However, this allows a wide range of evaluations: from development of characteristic species to generalized development of ecosystem-conditions. The evaluation of these probabilities needs knowledge of the target situation. If this target is not defined in terms of species, scarceness on national level, characteristic types of vegetation, etc., it is not possible to indicate the effects of the environmental change.

A simple example is given in Table 4 to illustrate the program. Four different realistic options are possible in the water management of a special area. The non-biotic conditions of the four options indicate an increasing salinity and an increasing eutrophication. Some concentrations and some results of these four sets of input are mentioned. If the optimal conditions of the species *Equisetum fluviatile* are required in the management, the conclusion is that the last option is worst, for it offers only a minimal percentage probability. Especially the first option with less nutrients and the most fresh conditions will stimulate this species. Evaluating the responses of 150 species, including their local importance, facilitates judgement on the general ecosystem development at the defined location.

Table 4 Some examples from four scenarios in ICHORS with different chemistry and the resulting responses of some species.

scenario no.	1	2	3	4
input data, e.g.				
chloride in mg/l	125	187	187	250
sulphate in mg/l	40	90	90	140
nitrate in mg/l	0.2	0.2	1.1	2.0
potassium in mg/l	8	8	16	25
species response, e.g.				
Equisetum fluviatile	0.622	0.393	0.233	0.037
Lemna trisulca	0.529	0.524	0.287	0.181
Chara vulgaris	0.349	0.493	0.006	0.000
Veronica beccabunga	0.231	0.209	0.144	0.001

The versions of ICHORS have been applied in the sequence of evaluations of the effects in changes in hydrology. According to the possibilities in water management in the mentioned regions, it is mostly a combined effect of changes in groundwater abstraction, changes in water tables and supply with river water (Barendregt et al., 1985, 1990, 1992, 1993; Wassen et al., 1986; Van Westrienen et al., 1991; Stam and Barendregt, 1992; Bootsma and Van Leerdam, 1993). These examples are from local to regional scale with interpolations and extrapolations of the ICHORS versions. Since the input of the program has to be defined properly, all these studies have to be combined with landscape ecological studies of the areas. A multiple application of ICHORS in the central provinces of the Netherlands will be explained in the next chapter.

7 ICHORS IN GROUNDWATER RESEARCH OF THE CENTRAL NETHERLANDS

The hydrology of the central parts of the Netherlands is characterized by the elevation, since a number of sandy ice-pushed Pleistocene hill-ridges are infiltration areas of precipitation, whereas the low-lying polders are the discharge areas of the groundwater from these hill-ridges. In these polders and in the gradients from the hill-ridges to the polders are many nature reserves depending on the groundwater from the hill-ridges. However, at many locations the groundwater in the hill-ridges is abstracted for drinking-water supply, so that the groundwater levels are lowered in the whole area (Figure 4). Moreover, the discharge into the polders decreased resulting in negative water balances and eutrophicated river water has to be supplied to fight the low water levels. Since there is a direct relation between water management and nature protection, the question arises if it is possible to reduce or re-arrange the locations of groundwater abstraction, so that the critical nature conditions are stimulated. It is not possible to solve this

GROUNDWATER MANAGEMENT IN CENTRAL NETHERLANDS

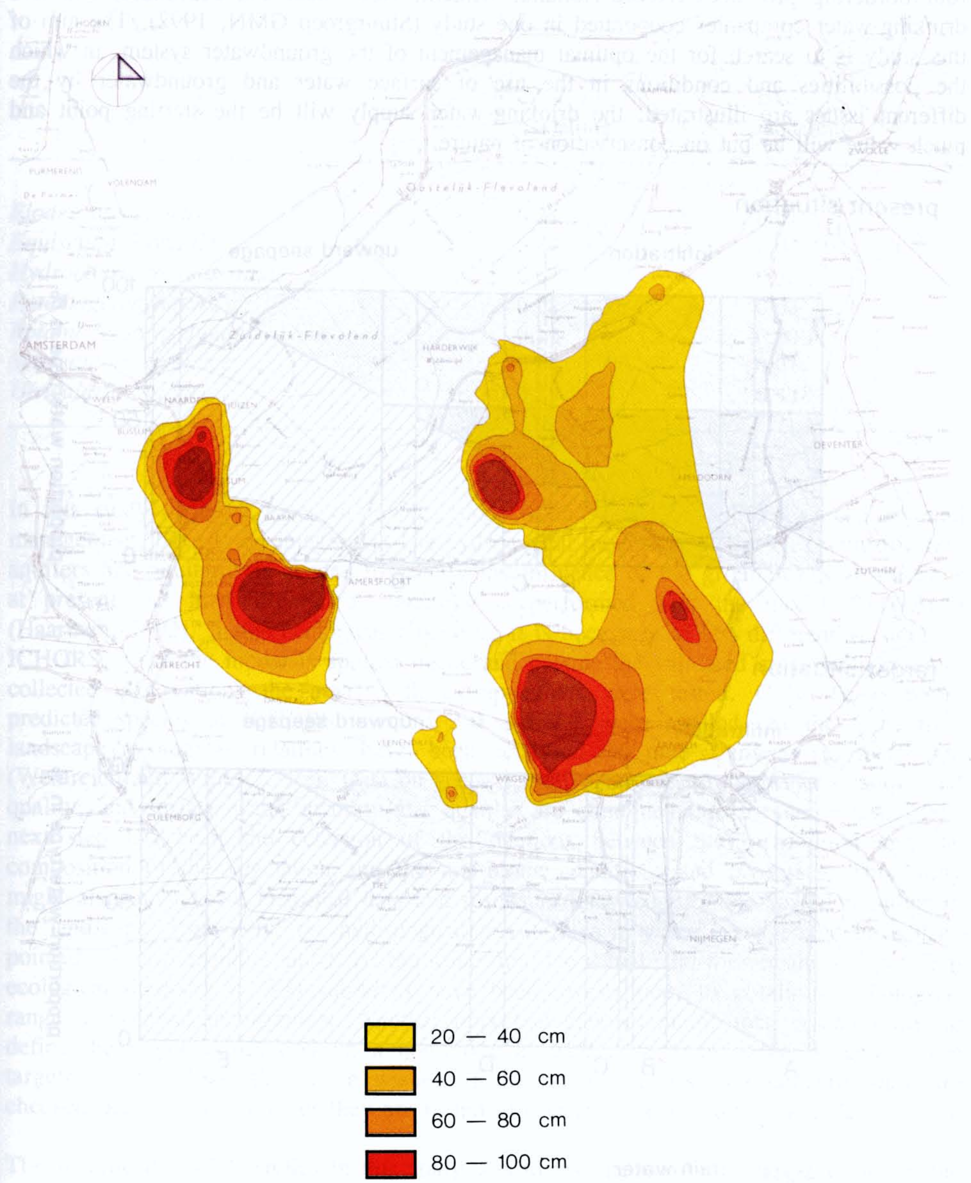


Figure 4 Effects of groundwater abstractions in 1985 on the piezometric levels in the study area of Groundwater Research of the Central Netherlands: lowering of groundwater levels in cm. From: Haarman, 1992. IWACO.

problem within one province, since the juridical borders do not coincide with the hydrological systems. For that reason, in an area of 70 km^2 in the central Netherlands four bordering provinces (Noord-Holland, Utrecht, Gelderland and Flevoland) and five drinking-water companies cooperated in one study (Stuurgroep GMN, 1992). The aim of this study is to search for the optimal management of the groundwater system, in which the possibilities and conditions in the use of surface water and groundwater by the different issues are illustrated; the drinking-water supply will be the starting point and much value will be put on conservation of nature.

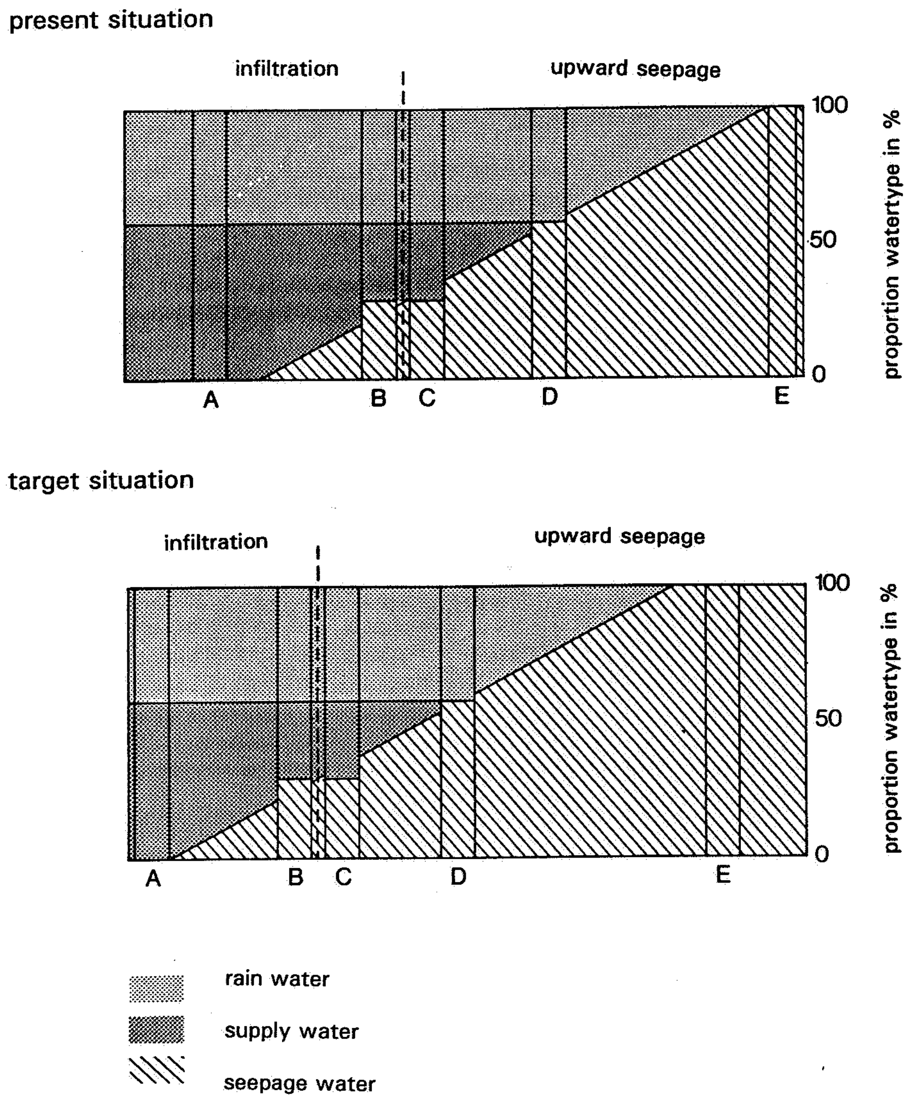


Figure 5 Spatial gradient with the proportion of seepage water, rain water and supply (river) water. A = supply water option; B and C = mixed options; D = seepage water option; E = intensive seepage option.

Table 5 Responses from ICHORS 3.0 with different physical dimensions in the subregion 209-211: Kortenhoef, and the same chemical and hydrological conditions.

	ditch-modelling	lake-modelling
Elodea canadensis	0.24	0.01
Equisetum fluviatile	0.84	0.08
Hydrocharis morsus-ranae	0.67	0.36
Potamogeton natans	0.28	0.03
Potamogeton trichoides	0.14	0.00
Sparganium emersum	0.14	0.03
Utricularia vulgaris	0.23	0.53

In this study different types of investigations in the fields of hydrology, ecology and management, have to be integrated. The hydrological modelling of the area with different aquifers and aquitards is needed to specify the influence of the groundwater abstractions at present and in future. This modelling is performed with the model TRIWACO (Haarman, 1992). In the ecological modelling it is necessary to test different versions of ICHORS, since the model is applied inside and outside the validated area. With recently collected observations the areas with extrapolations were tested. Only the correctly predicted species of ICHORS versions 3.0 and 3.2 were applied. At the same time landscape ecological relations have been analysed in nine transects of 6-15 km (Weinreich and Witjes, 1992). Data on geology, soil, land use, groundwater quantity and quality, and surface water quantity and quality, are combined into a relation scale. The next step has been the detection of the relations between the distribution and the composition of the vegetation and the non-biotic conditions and processes. This study might also incorporate historical data and results in knowledge of the spatial relations in the landscape. Especially, the hydrological possibilities in water management have been pointed out in a spatial context. At the same time the actual, the former and the potential ecological diversity in 300 sub-areas have been investigated, to obtain the ecological range in eventual management. Another important element in the total research was to define the target values both in a non-biotic as well as a biotic target situation. The targets are based on the vague wishes of the policy makers. Subsequently they are checked per sub-area whether they are technically visible and socially attainable.

The specific task of ICHORS in this study was to test the non-biotic target values in the management options for its ecological impact. The ecological judgement methodology was constructed by a number of steps. The hydrological options (in time and space) were tested on their ecological implications. Changing the hydrological options with regard to the present conditions would mean that in terrestrial conditions the level of phreatic water would rise and for that reason the area supplied with neutral nutrient poor groundwater would increase (Figure 5). In this way the area with upward seepage along the hill-ridges would expand. In the case of surface water, the proportion of exfiltrating groundwater next to the hill-ridges would increase and for that reason the proportion of groundwater in the surface water chemistry would increase; the proportion of river water (rich in salinity and nutrients; Barendregt, 1993) would decrease. These changes are estimated for each subregion individually since each subregion has its own characteristics. At that stage the model offered beneficial information about different conditions in the subregions. For instance in Table 5 some data are given from the area of Kortenhoef: with the same chemical composition of surface water quite different responses are given if the physical dimensions of the surface water system are changed.

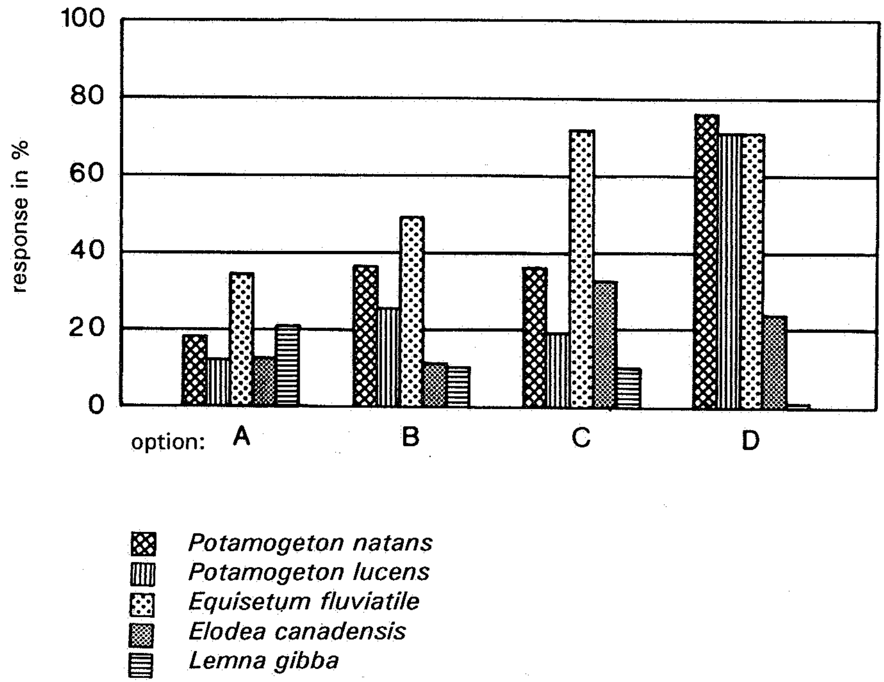


Figure 6 Responses of five species at different options in sub-area 2301/02 (Putterpolder). Options: see text.

Each subregion (or polder) was investigated with this method. Characteristic species were selected in each subregion and the data were evaluated. The responses were mentioned with four options in future:

A - supply with river water by infiltration;
B - mixed water from river and groundwater by infiltration;
C - mixed water from river and groundwater by upward seepage;
D - supply with groundwater by upward seepage.

Gentiana pneumonanthe, a threatened species of wet, acid and nutrient-poor sites

Tussock of *Scirpus cespitosus* in a heathland

Wet and relatively nutrient-poor meadow in a brook valley, with a.o. *Senecio aquaticus* and *Lychnis flos-cuculi*

Wet meadow with *Caltha palustris* in full bloom

GROUNDWATER MANAGEMENT IN CENTRAL NETHERLANDS

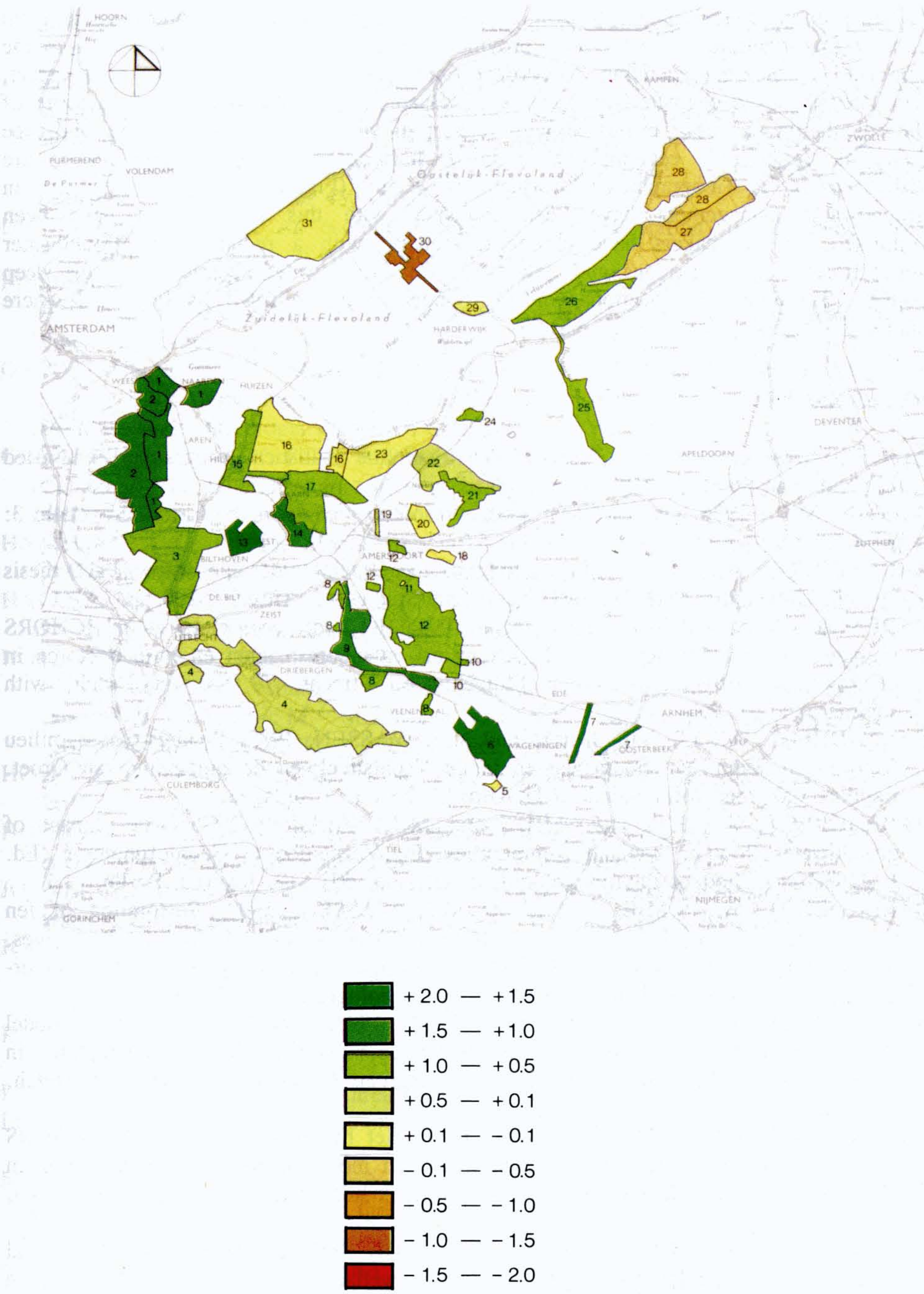


Figure 7 Scores of nature in 31 main regions from the evaluation of scenario no. 3.

To illustrate the different results in the options, the responses of some characteristic species in subregion 2301/02 (Putterpolder) are given in Figure 6. These values are compared with the highest possible scores with regard to the possibilities in management in the subregion, and obtain a score. These scores are multiplied with a value from another investigation, since the four provinces have given a value in each area to the presence of nature depending on groundwater. This value, in a four-point scale, facilitated to stress characteristic areas with valuable nature reserves. The final score of the calculation has been evaluated with respect to the other activities in the landscape (such as agriculture, economy, etc.) in a multi-criteria analysis. The results are visualized in a Geographical Information System (Figure 7). Five scenarios in groundwater management have been evaluated with this method and have been compared. Policy makers have chosen a scenario with a reduction of groundwater abstraction in the hill-ridges and new wells for groundwater abstraction in the deep polders. These are the locations where groundwater has fulfilled its flow and where nature is influenced for a minor part.

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DEMNAT: A NATIONAL MODEL FOR THE EFFECTS OF WATER MANAGEMENT ON THE VEGETATION

J.P.M. Witte, C.L.G. Groen, R. van der Meijden and J.G. Nienhuis

ABSTRACT

In the Netherlands, nature conservation and development is a relatively new aspect of water management policy. This article describes the ecohydrological model DEMNAT, designed to predict the impact of water management scenarios on groundwater-dependent vegetation types. Changes in four hydrological variables that can be predicted with existing hydrological models are used as input to DEMNAT. Outputs of the model are changes in the species richness of vegetation types and in resulting values for nature conservation. Apart from empirical calculation rules, the model contains a geographical schematization of the Netherlands in which information on nature, hydrology and soil is stored. Recently, DEMNAT was applied in a policy analysis on national groundwater management.

1 INTRODUCTION

The Netherlands is located in the delta of the rivers Rhine and Meuse. It is densely populated and heavily industrialized. Many vegetation types are phreatophytic because of the shallow groundwater table. In recent decades, the vegetation has suffered severely from the drawdown of the groundwater table caused by intensified drainage and groundwater extraction and from the pollution of ground- and surface water (Braat et al., 1989). Since nature has become a scarce commodity in the Netherlands, much attention has been paid to the natural environment in recent policy documents on water management. In order to include the natural environment explicitly in the analyses for these documents, the Dutch government developed the national ecohydrological model DEMNAT (Dose Effect Model for terrestrial NATure). With DEMNAT, the impacts of water management scenarios on the species richness of several terrestrial vegetation types can be predicted. In addition, DEMNAT evaluates these impacts in terms of nature conservation.

The first version of DEMNAT (Witte, 1990; Witte et al., 1992) was used for a policy document on national water management (Claessen, 1990). This paper describes DEMNAT-2 (Witte et al., in press), an improved version which was recently applied in the impact assessment for the National Policy Plan on Drinking Water and Industrial Water Supply (Beugelink et al., 1992). DEMNAT-2 is the product of close cooperation between scientists of the University of Leiden and the Wageningen Agricultural University, and research institutes of two ministries (the Institute for Inland Water Management and Waste Water Treatment (RIZA) of the Ministry of Transport, Public Works and Water Management and the National Institute of Public Health and Environmental Protection (RIVM) of the Ministry of Housing, Physical Planning and Environment).

DEMNAT makes predictions about the vegetation on the assumption that vegetation is the biotic ecosystem variable, affected most directly and severely by water management. The predictions relate to the medium long term (about 20 years), which is considered to be the relevant time horizon in policy making. Figure 1 gives the process chain of a prediction:

- Water management scenarios are implemented in hydrological models, which predict resulting changes in four hydrological variables.
- The hydrological changes affect abiotic soil factors which directly influence the species composition of the vegetation. These so-called 'operational' soil factors are (see e.g. Van Wirdum and Van Dam, 1984):
  - moisture regime (which is indirectly a measure for aeration);
  - nutrient availability;
  - acidity.

  The operational soil factors are conditioned by soil- and groundwater properties, such as soil texture, organic matter content, $CaCO_3$-content, groundwater level (Klijn et al., 1992).
- Changes in the operational soil factors result in changes in the species composition of the vegetation.
- Finally, the changes in species composition are expressed in nature values, to be used in nature conservation policies.

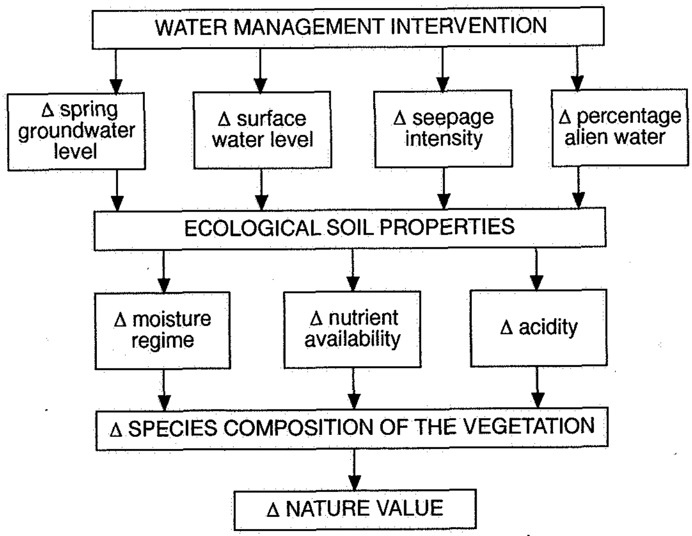

Figure 1 Conceptual process chain for a prediction by DEMNAT. Changes are indicated with a triangle (D).

This process chain implies that DEMNAT needs to contain the following modules:
1) a geographical schematization of the Netherlands with information on hydrology, soil and vegetation;
2) dose-effect functions for calculating the effects of water management measures on the vegetation;
3) a procedure to evaluate these effects in terms of nature conservation.

In the next Section we will go into the hydrological input for DEMNAT. Then we will describe the respective modules of the model in Section 3, 4 and 5 respectively. To demonstrate DEMNAT's output, some simulation results will be given in Section 6. Finally, in Section 7 we will discuss DEMNAT's usefulness, show its main shortcomings and mention possible improvements.

2 HYDROLOGICAL INPUT

In contrast to many other models, such as WAFLO (Gremmen et al., 1990), ICHORS (Barendregt and Nieuwenhuis, 1993) and MOVE (Latour et al., 1993), DEMNAT uses hydrological changes as input, and not absolute values. Changes can be predicted more reliably by hydrological models than absolute values, because systematic errors are minimised when calculating a difference. Input to DEMNAT are changes in hydrological variables, which can be predicted with existing hydrological models and which are decisive for the plant species composition of the vegetation. These are changes in (see Figure 1):
1) Spring Groundwater Level (*SGL*);
2) surface water level;
3) seepage intensity;
4) percentage alien water in summer, i.e. water from the rivers Rhine and Meuse.

Existing hydrological models are linked to DEMNAT in such a way that their outputs can be used directly as input. The first two variables are calculated with the Groundwater Model for the Netherlands (LGM) per grid cell of 1 × 1 km for six so-called groundwater classes (Pastoors, 1992). A groundwater class indicates the depth of the groundwater table below ground surface. Change in seepage intensity (3) is also calculated with LGM per square km, however not differentiated to groundwater classes. The term 'seepage' refers to the groundwater flow from the top aquifer to the phreatic groundwater. The change in percentage alien water (4) is calculated with a surface water distribution model for calculation units which vary in size from dozens to hundreds of km^2 (Pakes et al., 1992).

The first variable influences only terrestrial vegetation types; the second influences only semi-aquatic vegetation types, i.e. vegetation types of terrestrializing sites and small water bodies, such as ditches. Variables 3 and 4 influence both terrestrial and semi-aquatic vegetation types.

3 GEOGRAPHICAL SCHEMATIZATION OF THE NETHERLANDS

3.1 Vegetation

Distribution maps of vegetation types were derived from information concerning the distribution of vascular plants that grow in the 'wild'. This information is stored in a data bank (FLORBASE) that recently became operational (Groen et al., 1992). FLORBASE contains per grid cell of 1 × 1 km, data on the presence of indigenous plant species in the period 1975-1990; it does not contain more detailed information within a square kilometre, for instance about the abundance of plant species. Indicator species were used to determine the relative plant species richness or completeness of vegetation types. The vegetation typology used is a slightly modified version of the ecosystem classification developed by the Centre of Environmental Science (Stevers et al., 1987). In this classification, small-scale ecosystems ('ecotopes') are distinguished using vegetation structure and the operational soil factors 'moisture regime', 'nutrient availability' and 'acidity' as differentiating characteristics.

Table 1 describes the 15 vegetation types taken into account. These are all worthy of nature conservation as well as susceptible to water management operations. Each vegetation type is coded by three symbols, which give information about the characteristics used to define the vegetation type. A capital indicates the structure of the vegetation (A=semi-aquatic, K=herbaceous vegetation, H=woodland and shrub), the first number refers to the moisture regime (1=semi-aquatic, 2=wet, 4=moist), the second number to both nutrient availability and acidity (1=nutrient poor and acid, 2=nutrient poor and moderately acid to neutral, 3=nutrient poor and basic, 7=moderately nutrient-rich, 8=nutrient-rich).

Table 1 Description of the vegetation types used in DEMNAT-2.

Code	Description
A12	Vegetations in stagnant, nutrient-poor, moderately acid to neutral freshwater
A17	Vegetations in stagnant, freshwater of moderate nutrient availability
A18	Vegetations in stagnant, nutrient-rich freshwater
K21	Herbaceous vegetations on wet, nutrient-poor, acid soil
K22	Herbaceous vegetations on wet, nutrient-poor, moderately acid to neutral soil
K23	Herbaceous vegetations on wet, nutrient-poor, basic soil
K27	Herbaceous vegetations on wet soil of moderate nutrient availability
K28	Herbaceous vegetations on wet, nutrient-rich soil
K41	Herbaceous vegetations on moist, nutrient-poor, acid soil
K42	Herbaceous vegetatations on moist, nutrient-poor, moderately acid to neutral soil
H22	Woodlands and shrubs on wet, nutrient-poor, moderately acid to neutral soil
H27	Woodlands and shrubs on wet soil of moderate nutrient availability
H28	Woodlands and shrubs on wet, nutrient-rich soil
H42	Woodlands and shrubs on moist, nutrient-poor, moderately acid to neutral soil
H47	Woodlands and shrubs on moist soil of moderate nutrient availability

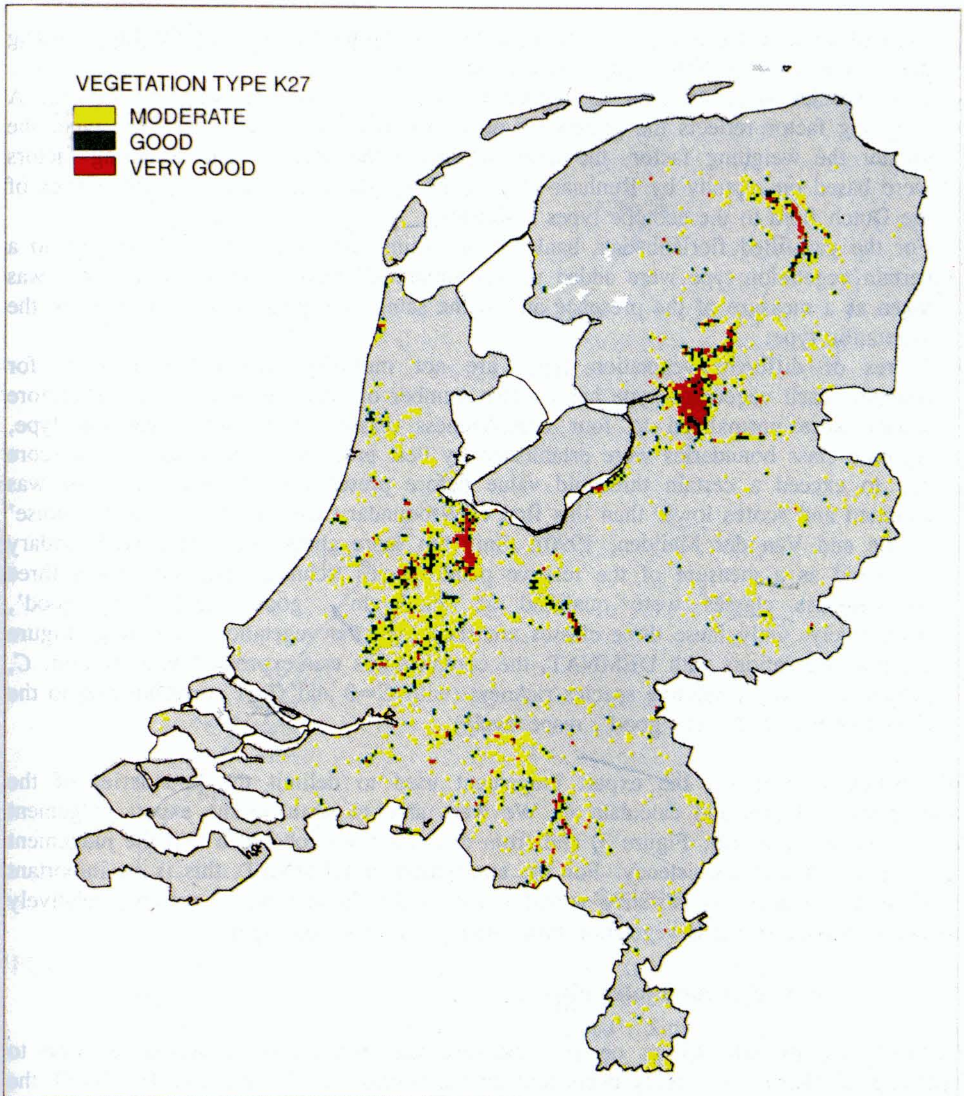

Figure 2 Distribution map of vegetation type K27: 'herbaceous vegetations on wet soil of moderate nutrient availability' (Witte and Van der Meijden, 1992).

Figure 2 shows the distribution map of vegetation type K27 (herbaceous vegetation (K) on a wet (2) soil of moderate nutrient availability (7)). From FLORBASE, Witte and Van der Meijden (1992; in press) derived this map and maps of the other vegetation types in Table 1 as follows:

- FLORBASE cannot be used directly for the construction of vegetation maps, because the intensity of plant inventories in the Netherlands differs from region to region. So, in order to make reliable maps a gap-filling method was applied to FLORBASE to obtain a second, more complete, data bank. This gap-filling method is based on calculated correlations between plant-records from well-investigated regions. On the

basis of these correlations it is possible to identify with a high probability, missing plant records in less-thoroughly investigated regions;
- Plant species were ascribed to vegetation types by means of weighing factors. A weighing factor reflects the preference of a species for a certain vegetation type: the greater the weighing factor, the more indicative the species. The weighing factors were based on a study by Runhaar et al. (1987), who ascribed each plant species of the Dutch flora to the ecotope types it prefers;
- For the gap-filled floristic data bank, all weighing factors of species belonging to a certain vegetation type were added up per square kilometre. The resulting score was taken as a measure of the presence and at the same time plant species richness of the vegetation type;
- Scores of different vegetation types are not mutually comparable because, for instance, each vegetation type has its own number of characteristic species. Therefore scores were normalized to four completeness classes. For each vegetation type, separate class boundaries were established by best professional judgement. The score had to exceed a certain threshold value before presence of a vegetation type was assumed and scores lower than this first class boundary were considered to be 'noise' (Witte and Van der Meijden, 1990). Only the score above the first class boundary was used as a measure of the relative plant species richness. The subsequent three completeness classes were qualified as 'moderate', 'good', and 'very good', respectively. Only these three classes are shown on the vegetation maps (e.g. Figure 2). For calculations with DEMNAT, the completeness was expressed as a fraction, C, proportional to the relative species richness (with C=0 and C=1 corresponding to the classes 'noise' and 'very good', respectively).

We critically examined the expert judgement used to delimit the boundaries of the completeness classes and calculate C. We were able to simulate the expert judgement very accurately (see e.g. Figure 3) and from this result we concluded that the judgement has a great internal consistency. For the application in DEMNAT, this is an important conclusion, because inconsistency would result in the domination of regions, relatively abundant in overestimated vegetation types (that is, with C too high).

3.2 Soil and groundwater class

DEMNAT needs information on soil, because the reaction of a vegetation type to hydrological changes is largely dependent on controlling soil factors. For DEMNAT the more than 2 000 units of the 1 : 50 000 Soil Map of the Netherlands were aggregated to 52 ecologically relevant units in two steps by respectively De Waal (1992) and Klijn et al. (1992).

The doses 'change in spring groundwater level' and 'change in surface water level' are predicted with LGM per square kilometre for 6 groundwater classes (see Section 2). The geographical information on groundwater classes, needed by DEMNAT to use this output directly, was provided by Klijn et al. (1992).

The preference of a certain vegetation type for each combination of ecological soil unit and groundwater class was given semi-quantitatively in the form of an estimated probability P (Klijn et al., 1992). This estimation refers to undisturbed conditions. Fifteen vegetation types linked to 52 ecological soil units and 6 groundwater classes yielded 4 680 elements in a probability matrix. The probabilities are used in DEMNAT for spatial downscaling of the information on vegetation types per square kilometre (see next Section).

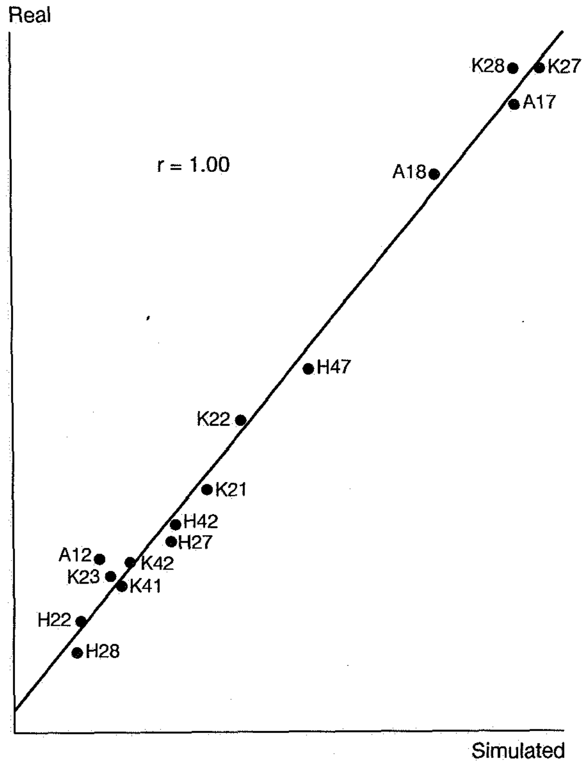

Figure 3 Simulated values of the third class boundary (x-axis) put out against the real expert judgement (y-axis). The third class boundary divides scores in the completeness classes 'good' and 'very good'. Similar results were obtained for the first and second boundary class. (Witte and Van der Meijden, 1992).

3.3 Combining geographical information: the ecoplot

To achieve homogeneous calculation units, an overlay was made of the vegetation maps, the ecological soil map and the groundwater class map (Nienhuis, in press). These units or ecoplots are stored in a file which is read sequentially when making a computation. DEMNAT will give its results per ecoplot. These results can later be aggregated to results per vegetation type, or per square kilometre, for instance. The use of ecoplots provides readily accessible information and has the advantage to limit the computation time (60 minutes for a scenario on a HP-workstation).

Each ecoplot is described by number codes for the vegetation type and the ecological soil unit; codes for the location of the square kilometre and for the groundwater class give access to the hydrological inputs, coming from the hydrological models. But there is more information, used in the computation, attributed to each ecoplot (Witte et al., in press):

- The potential area A_{pot} of the vegetation type, that is the presumed size in undisturbed conditions. A_{pot} is used in the nature valuation and obtained by multiplying the ecoplot area with the probability of the vegetation type P;
- The fraction of the vegetation type that can be ascribed to the ecoplot. The ecoplot results have to be multiplied with this fraction in order to count each vegetation type just once, in spite of its presence in more than one ecoplot within the square kilometre. To calculate this fraction, we assumed that each vegetation type is distributed over its associated ecoplots proportionate to their potential areas A_{pot}.

If, for instance, a square kilometre consists of three ecoplots of 10, 40 and 50 ha, and the probabilities P containing a certain vegetation type X are 0.10, 0.50 and 0.08 respectively, then the potential areas A_{pot} of X in these ecoplots will be 1, 20 and 4 ha. The fractions of X that can be ascribed to each ecoplot are in this case 0.04, 0.80 and 0.16.

4 EFFECTS OF HYDROLOGICAL CHANGES ON THE VEGETATION

DEMNAT uses separate dose-effect functions for each possible combination of ecological soil unit and vegetation type. These functions were calculated in two steps, see Figure 4 (Van der Linden et al., 1992; Runhaar and Van der Linden, in press). First, it is determined which changes in operational soil factors are expected to occur, given a certain dose (i.e. hydrological change). Second, empirical relationships between species composition and operational soil factors are used to predict how these changes will affect the vegetation.

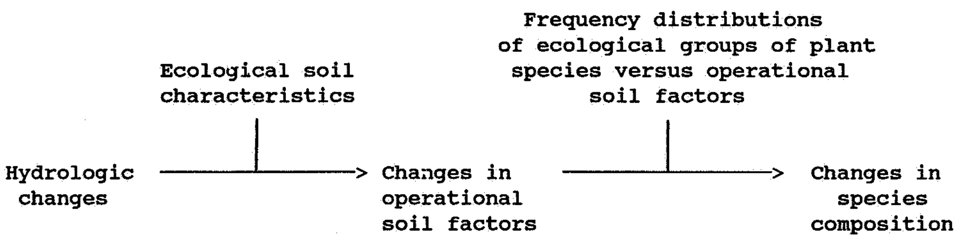

Figure 4 Calculation of dose-effect functions (free after Van der Linden et al., 1992).

Data from our own measurements and literature (see Van der Linden et al., 1992) were used to estimate the effect of a dose on the operational soil factors (first step), characterized by:

- moisture regime:
  - spring groundwater level (applied to terrestrial vegetation types)
  - surface water level (semi-terrestrial)
- nutrient availability:
  - N-mineralisation (terrestrial)
  - phosphate concentration (semi-terrestrial)
- acidity:
  - pH of the soil (terrestrial)

Extrapolation of the limited empirical data to other ecological soil units was necessary in order to obtain the effects of these variables for all 52 units. The effect on the vegetation (step 2) was calculated with the help of frequency distributions of ecological groups of plant species versus operational soil factors. Using these frequency distributions as response curves, the effect on the vegetation of changes in operational soil factors could also be calculated.

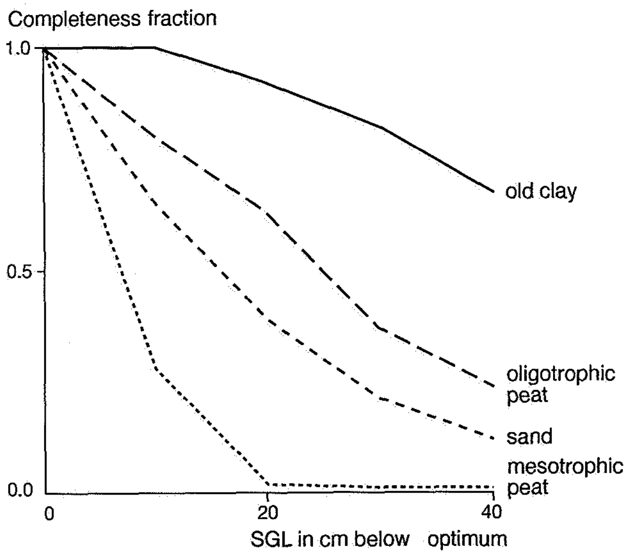

Figure 5 Example of some dose-effect functions for a lowering of the *SGL* (viz. vegetation type K21: herbaceous vegetation on wet, nutrient poor, acid soil) (Van der Linden et al., 1992).

Figure 5 shows the response of vegetation type K21 (herbaceous vegetation on wet, nutrient-poor, acid soil) to a drawdown of the groundwater ($dSGL<0$) for four ecological soil units. Similar functions were established for the other doses: 'change in surface water level', 'change in seepage intensity' and 'change in percentage alien water'. The use of these functions is demonstrated in Figure 6A. This figure shows that the original completeness fraction is used to estimate the initial *SGL*, that is the *SGL* before any hydrological change has taken place. A sensitivity analysis showed that DEMNAT is rather robust for this interpretation (Witte, 1990, p. 52).

Policy makers also want to be able to predict the recovery of vegetation types as a result of favourable water management measures. Predicting recovery, however, requires additional geographical information, e.g. on vegetation management and the supply of nutrients, information which is not yet available on national scale. For the prediction of recovery we temporarily introduced hysteresis factors which reduce the slope of the original dose-effect functions (see Figure 6B). These factors, based on best professional judgement, depend on the type of vegetation and relate to the prediction term of 20 years. Because recovery asks for the availability of diaspora, we assumed that a certain amount of characteristic plant species must be present before the hysteresis factor can be applied. In practice this means that the completeness fraction, *C*, must exceed zero.

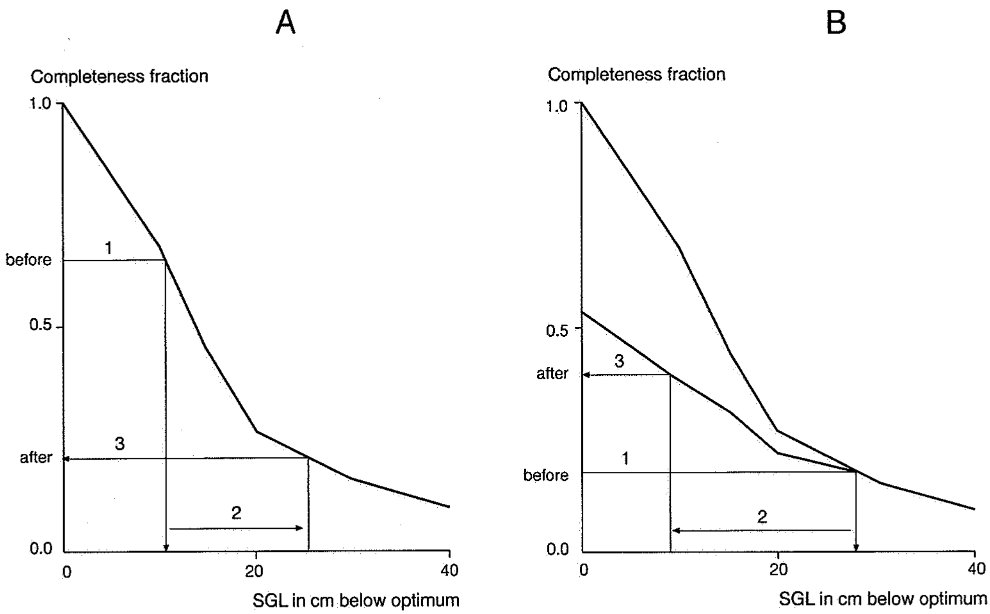

Figure 6 Use of the dose-effect functions: (A) with a fall in the *SGL*; (B) with a rise in the *SGL* (Witte, 1990). First, the initial *SGL* is estimated from the original completeness fraction (1). The predicted change in groundwater level, d*SGL*, is then added to this initial value, resulting in a new *SGL* (2). Finally, the completeness fraction after water management measures is found (3).

Van der Linden et al. (1992) were able to validate some of the functions for a lowering of the groundwater table ($dSGL < 0$) based on known effects described in literature. The predicted effects were largely consistent with the observations.

An important difference between DEMNAT and other ecohydrological models (WAFLO, ICHORS, MOVE) is that the initial soil conditions (such as moisture regime) are derived from the vegetation itself. In our opinion far better information about soil conditions can be obtained from the vegetation than from any other available data bank or model. For instance, vegetation type K21 indicates a wet, nutrient-poor and acid soil (see Table 1).

Furthermore there is a difference between the functions statistical models (ICHORS, MOVE) use and those that DEMNAT uses for a recovery. Although we do not consider the introduction of hysteresis factors as a final solution, we believe that we should take into account the fact that it takes vegetation longer to recover than to deteriorate.

5 VALUATION OF EFFECTS

If desired, the changes in the completeness fractions of the vegetation types can be expressed in changes in nature values. Nature valuation enables the outcomes for different vegetation types to be combined, thus yielding results that are easy to interpret. Because the valuation of nature is a subjective affair, we strictly separated the prediction of effects on the completeness fraction from the assessment of these effects in terms of nature conservation. Moreover, the user of DEMNAT is able to change the valuation procedure to some extent.

In brief, the nature valuation procedure presently implemented is as follows (Witte and Van der Meijden, 1992):

The potential nature value for each vegetation type, V_{pot} (Table 2) was determined from the criteria 'national rarity' and 'international rarity' once only. This value refers to floristically very well developed representatives of the vegetation type in question. The actual value of a vegetation type in a certain grid cell is calculated from C, V_{pot}, and a weighing factor for the area of the vegetation type within the cell, F, which is a function of the potential areas A_{pot}:

$$V_{act} = V_{pot}FC \qquad (1)$$

where: V_{act} Actual nature value
V_{pot} Potential nature value
F Weighing factor for the vegetation area
C Completeness fraction

The gain or loss of nature values due to predicted changes in the completeness fraction, dC, is simply calculated as:

$$\mathrm{d}V_{act} = V_{pot}F\mathrm{d}C \qquad (2)$$

Adding all actual nature values of the vegetation types within one grid cell yields the nature value of groundwater-dependent vegetation types in that grid cell.

Table 2 Potential nature values, V_{pot}, for the vegetation types described in Table 1.

Vegetation type	Value	Vegetation type	Value	Vegetation type	Value
A12	31.7	K23	20.0	H22	16.4
A17	2.5	K27	1.5	H27	2.2
A18	1.2	K28	1.0	H28	3.8
K21	5.9	K41	2.2	H42	4.1
K22	6.7	K42	6.0	H47	2.3

6 SIMULATIONS

DEMNAT-2 has been applied in a national policy analysis for the benefit of a document on drinking-water and industrial-water supply (Beugelink et al., 1992). One of the scenarios analysed was about a 50% decrease of the drinking-water abstraction. In this Section some results are given for a certain region of the Netherlands, the province of Utrecht (Figure 7). Figure 8 and 9 show the hydrological input of this scenario for DEMNAT. Figure 8 gives the effect on the *SGL*, averaged per square kilometre. A substantial rise of the *SGL* can be expected in an ice-pushed ridge with deep groundwater levels, lying roughly across the figure from SE to NW. Especially at the edge of the ice-pushed ridge, intensities of upward seepage will increase (Figure 9). The effect of this scenario on surface water quality is almost negligible (Pakes et al., 1992, p. 8).

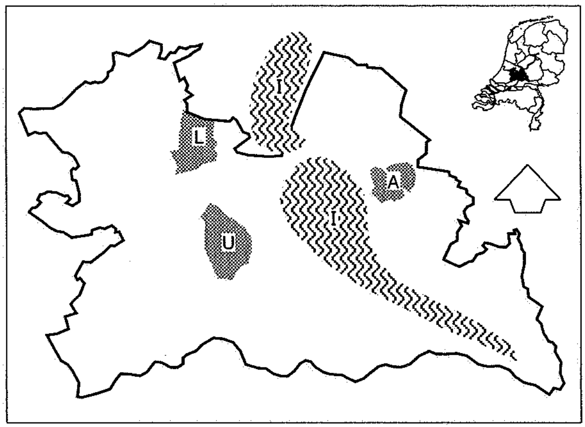

Figure 7 Province of Utrecht with: (A) city of Amersfoort; (I) ice-pushed ridge (roughly indicated); (L) lake Loosdrecht; and (U) city of Utrecht.

The ecological impacts of these hydrological changes were predicted with DEMNAT. Figure 10 gives one of the results, the effect on the nature value per grid cell. A substantial increase in nature values may be expected at the edge of the ice-pushed ridge. No effect will occur on the ridge itself. Some groundwater-dependent vegetation types are present on the ridge, but they only occur on soils with a perched water table. As these soils cannot be influenced by water management, DEMNAT ignores these occurrences.

Figure 11 shows another example of DEMNAT's output, which is the result of a completely fictive scenario: an overall lowering of the groundwater and surface water level with 10 cm, without any change in upward seepage or surface water quality. In this case, dramatic effects are predicted, especially for the western part of the province of Utrecht. A highly affected region can be seen at the west side of the ice-pushed ridge. This region is well known for its groundwater- and upward seepage-dependent vegetation (especially K22). Figure 12 shows the effect on the average completeness fraction in the province of Utrecht for all the vegetation types. On average, the

Mown reed land in spring with the purple *Lychnis flos-cuculi*

Workers removing a layer of *Sphagnum* moss in a mown reed land

A ditch, characteristic of the Dutch polder landscape, invaded by *Stratiotes aloides* and *Potamogeton natans*

A pond in a peatland with *Nymphaea alba,* a very common species in the Netherlands

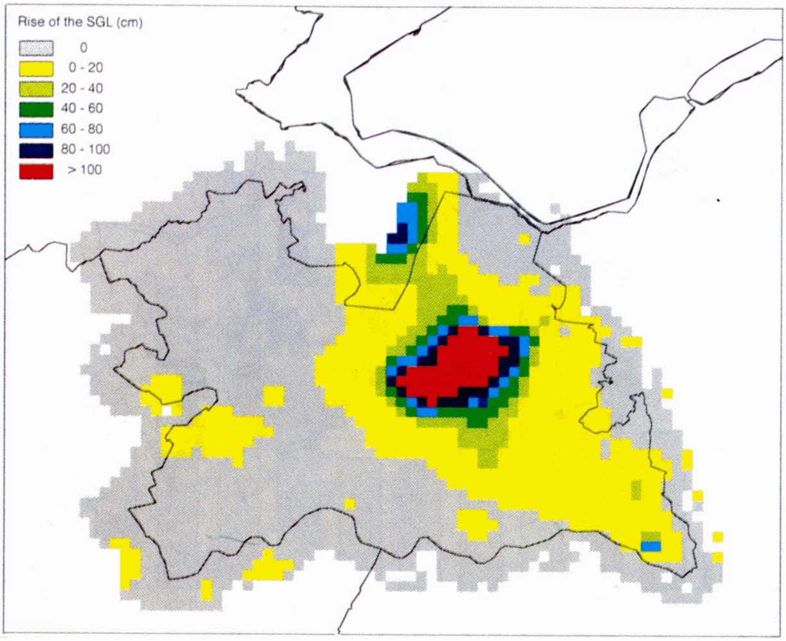

Figure 8 Effect on the main Spring Groundwater Level of a 50% decrease of the drinking-water abstraction. Calcualted with LGM (Pastoors, 1992).

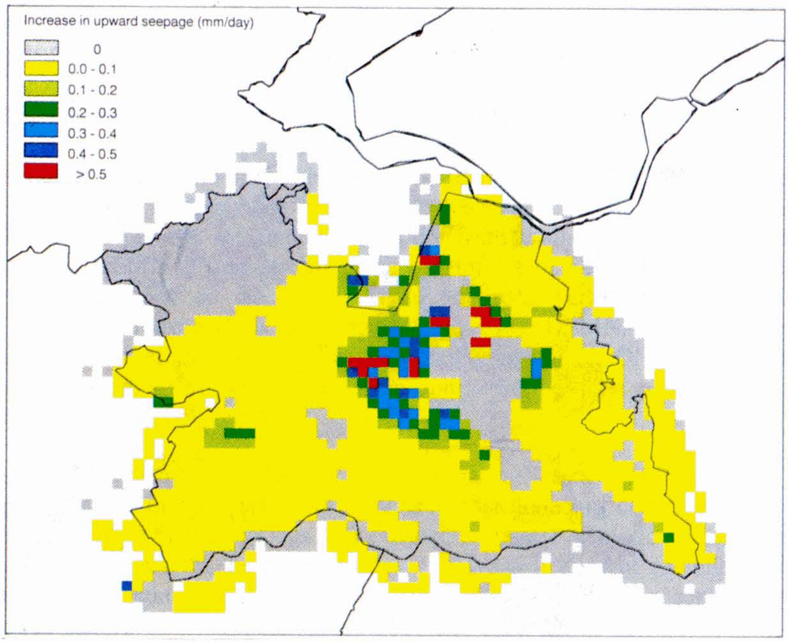

Figure 9 Effect on upward seepage intensities of a 50% decrease of the drinking-water abstraction. Calculated with LGM (Pastoors, 1992).

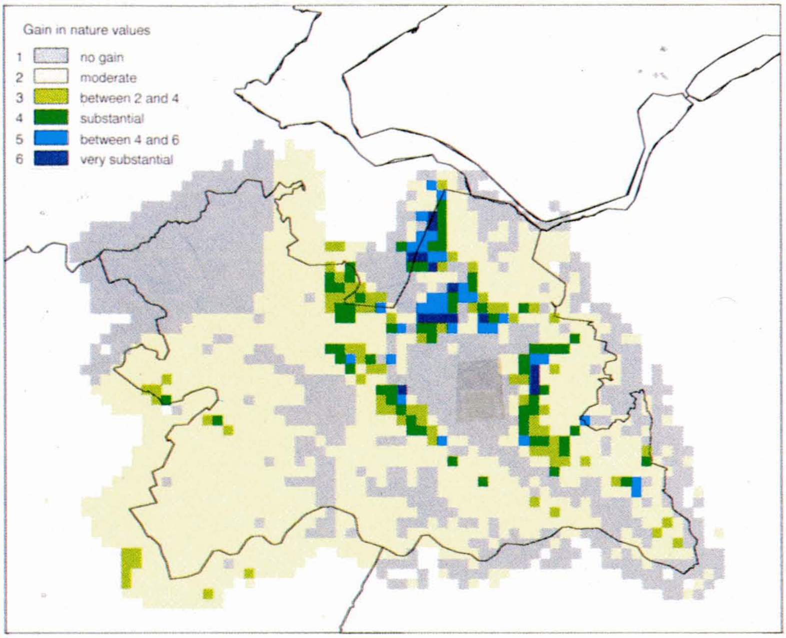

Figure 10 Change in nature values as a result of a 50% decrease of the drinking-water abstraction. Calculated with DEMNAT (Beugelink et al., 1992).

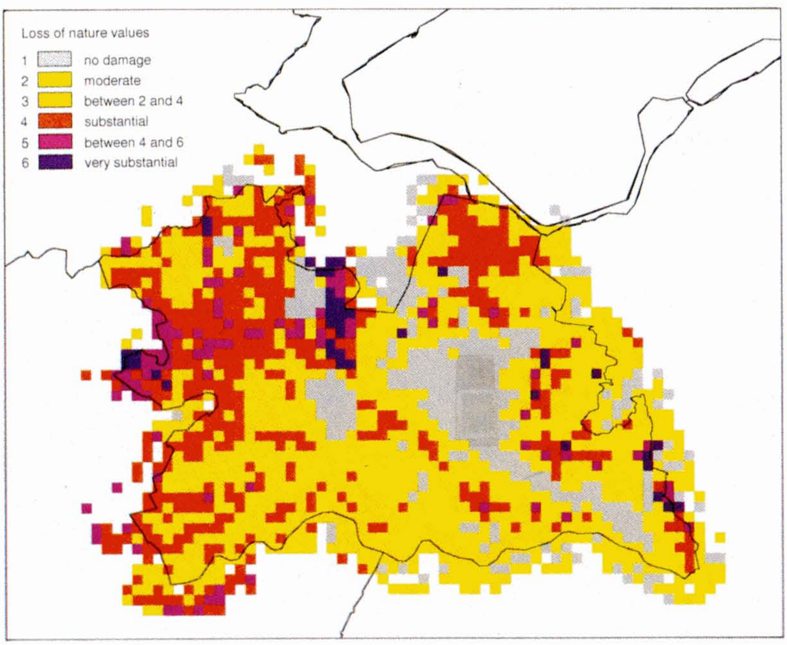

Figure 11 Predicted change in nature values as a result of a 10 cm fall in groundwater level and surface water level (Witte et al., in press; Nienhuis, in press).

vegetation types will decline with some 25%, but for vegetation type K22 the predicted decline is almost 60%.

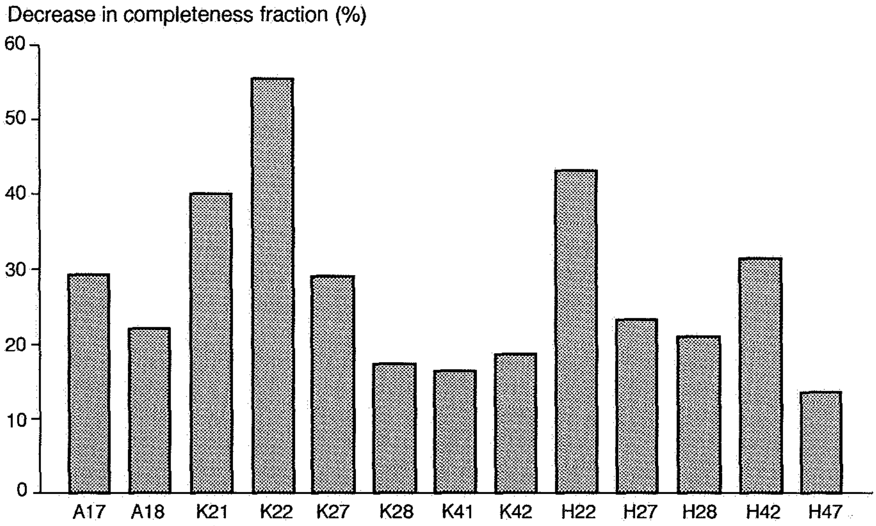

Figure 12 Predicted decrease in the mean completeness fraction in the province of Utrecht for all the vegetation types as a result of a 10 cm fall in groundwater level and surface water level (Witte et al., in press).

7 DISCUSSION

7.1 Introduction

DEMNAT's power lies in the fact that the model comprises all the elements that are needed for a prediction: geographical information, dose-effect functions, and a valuating system. Another plus point is the smooth connection with available hydrological models. In policy analyses DEMNAT has proved to be a helpful tool. As with any model, one can ask whether an expert on the subject would not make better predictions. Though answers to this question are hard to validate, we believe that DEMNAT has now reached a state in which it makes the use of experts at least superfluous. Of course DEMNAT also has its shortcomings. We will discuss them and show the main possibilities for the improvement of DEMNAT in Section 7.3.

Although we have made a number of comparisons between DEMNAT and other ecohydrological models in this paper, we must emphasize one big difference with other models: the use of plant species groups (viz. vegetation types) as biotic variables instead of individual plant species. In the next Section (7.2) we will account for this approach. For a comprehensive comparison of ecohydrological models, see the contributions of Claessen (1993, this volume) and Garritsen (1993, this volume).

7.2 Plant species groups or individual plant species?

In contrast to DEMNAT, most ecohydrological models make predictions for individual plant species. Plant species have the advantage of being clearly distinguishable objects, which are not - as groups of plant species - dependent on a more or less arbitrarily chosen typology. However, we believe that the use of plant species in a national model has more disadvantages:

- Errors in a floristic data bank directly affect the results when individual plant species are used. We were able to minimise these errors by taking groups of species, introducing threshold values, and by gap-filling per vegetation type (see Witte and Van der Meijden, 1992; in press);
- The dynamics in the appearance and disappearance of plant species is much bigger than the dynamics of ecological groups as a whole. This probably explains the moderate results of predictions for individual plant species (cf. Gremmen et al., 1990, p. 152);
- For easily manageable results, nature valuation is a necessity. Nature valuation on the basis of plant species gives an underestimation of species-poor ecosystems, such as bogs. Moreover, such a valuation results in giving a higher value to, for instance, a heather with nettle (*Urtica dioica*) than a heather without it. Therefore, we believe it is better to valuate on the basis of the relative richness of plant species groups (as expressed for DEMNAT in the completeness fraction C);
- For a reliable prediction, the involvement of spatial variability in soil conditions is a prerequisite. We incorporated this variability in DEMNAT by linking 15 vegetation types to 52 ecological soil units and 6 groundwater classes, thus yielding a probability matrix of 4 680 elements. If we would use 700 plant species (MOVE), the number of possible combinations to be made would be 218 400. We seriously doubt if there is enough knowledge to perform this operation;
- National applicable dose-effect functions for individual plant species can only be derived from an extensive data bank with a representative set of records of soil characteristics and plant species. At this moment, such a data bank is not available. We think the method described by Latour et al. (1993) is reprehensible, because the data bank with vegetation relevées they use (Wiertz et al., 1992) is anything but representative (Van der Meijden, 1993). Moreover, we do not think it is allowed to use average Ellenberg-indicator values of vegetation relevées for the construction of species-response curves, because averaging does not account for the fact that vegetation relevées often contain a broad spectrum of micro-sites (see e.g. Van Wirdum, 1991).

7.3 Shortcomings and possibilities for improvements

DEMNAT may be compared to a large cathedral with some pillars made of granite and others of clay. Clearly, improvements are possible and required to get a model that is solid in all its components.

In order to get a working model, DEMNAT had to be based on a number of important assumptions. For instance:

- we assumed that the occurrences in a grid cell of individual plant species can be used to assess vegetation types (Section 3.1);
- the maps on soil types and on groundwater classes were used to downscale the information on vegetation types (Section 3.3);
- we used the completeness fraction to estimate the initial *SGL* (Section 4);
- we estimated hysteresis factors for the recovery of vegetation types (Section 4).

Unfortunately, some assumptions might be hard to validate because of a shortage of necessary field data. However, to know how the assumptions influence the results, we shall at least perform a sensitivity analyses in the near future.

Presently, DEMNAT is not able to make predictions for certain regions because of a shortage of floristic information in FLORBASE. For a period of two years we intend to explore new sources of floristic observations, for instance those which are presently stored in an analogous form only. For well investigated grid cells we will compare the vegetation types that we derived from FLORBASE with vegetation types assessed from real vegetation mapping (Groen and Witte, in press). From this comparison we expect to find a function for the vegetation area, depending on a.o. the completeness fraction C and the potential area A_{pot}. This new function might be used in the nature valuation module. Another improvement in the short term can be the combination of DEMNAT with other hydrological models. At this moment the research institute RIZA is working on a combination of the national groundwater model NAGROM with a dynamic model for the unsaturated zone, DEMGEN. When these models are coupled to a surface water distribution model, it is possible to calculate the influence of alien water more accurately.

Substantial improvements of the dose-effect functions can only be established if more systematic data of vegetations together with soil characteristics become available. At present, there are not enough reliable data to validate the dose-effect functions (Gremmen et al., 1990), and therefore systematic monitoring programmes, running for a period of many years in carefully selected sites, are urgently needed. Another improvement in the long term might be the use of other geographical data banks, for instance on nutrient supply.

Perhaps in the near future DEMNAT will undergo a completely different kind of 'improvement'. At present DEMNAT is only suitable for analysing scenarios with respect to water management, especially those scenarios that relate to 'drought damage'. However, it may also become suitable for analysing two other environmental stressors: acidification and eutrophication. As the current DEMNAT-version makes predictions via the operational factors 'soil moisture regime', 'nutrient availability' and 'acidity', this adjustment should be relatively straightforward.

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MOVE: A MULTIPLE STRESS MODEL FOR VEGETATION [1)]

J.B. Latour, R. Reiling and J. Wiertz

ABSTRACT

In this contribution a multiple stress model for vegetation (MOVE) will be reviewed, which predicts the effects on ecosystems of stresses imposed by different environmental problems. The model consists of two modules: one for soil and one for vegetation. In the soil module changes in abiotic site factors (e.g. soil moisture, soil nutrient availability and soil acidity) are predicted for various environmental scenarios. In the vegetation module the ecological response functions of species (plotted in a multi-dimensional hypervolume according to Hutchinson) are used to predict the occurrence probability of species for each combination of the abiotic site factors. By linking the two modules, the occurrence probability of species can be predicted in each geographical unit. The model can be used for scenario studies, standard-setting and the identification of key problems and problem areas. Examples given are based on specific pilot studies.

1 INTRODUCTION

Changes in ecosystems, populations and species are often attributed to combinations of environmental stresses (e.g. Baldock, 1990). Studies focusing on one stress exclusively may have outcomes with confined reality if other stresses interfere. Studying the combined effect of stresses is complex (Latour and Reiling, in press, a.) because of regional and temporal variation in types and intensity of stresses, sensitivity of soils (buffering capacity) and vulnerability of ecosystems, as well as our limited understanding of the "pathology" of combination stress.

Most models that relate to more than one environmental problem are functionally mechanistic. These models reflect a trade-off between the geographical scale of the model, the number of ecosystems referred to and the complexity of processes described. We question the possibility of covering the many different ecosystems and various environmental problems on a national scale with functionally mechanistic models.

1) This paper was also presented at the symposium "Ecosystem Classification for Environmental Policy and Conservation", in Lunteren, The Netherlands, 17-18 December 1992.

Alternatively, multiple stress modelling may be possible using the concept of risk assessment. Risk assessment (VROM, 1991) is used for various environmental problems to quantify ecological limit values that correspond to similar protection levels for ecosystems. Risk assessment is based on probabilistic rather than deterministic analysis, and can be elaborated for various environmental problems. Risk assessment has been used in the field of ecotoxicology to assess concentrations of various compounds protecting a standardized percentage of the species of an ecosystem. Similarly, Van der Eerden et al. (1992) calculated critical levels for NH_3 and SO_2 using a risk assessment. Latour and Reiling (1992) adopted the risk assessment for calculation of eutrophication, acidification and desiccation.

Latour and Reiling (1991) developed a conceptual, species-centred, multiple stress model for vegetation (MOVE) that uses the concept of risk assessment. The model predicts the occurrence probability of individual species as a function of environmental variables, such as nutrient availability, soil pH and soil moisture. Simulation of chemical cycling in soil and soil moisture is used to assess changes in site factors.

In this paper we will review the model MOVE, pointing out its use for multiple stress modelling on a national scale and highlighting such applications as prediction, determination of ecological limit values and the ranking of environmental threats. The model has not yet been completed. Examples given are based on specific pilot studies.

2 MULTIPLE STRESS MODEL, MOVE

The Multiple stress mOdel for VEgetation (MOVE) (Latour and Reiling, 1991) predicts probability of occurrence of individual plant species as a function of acidification, desiccation and eutrophication scenarios on a national scale. The model consists of a soil module predicting the abiotic effect and a vegetation module predicting the corresponding biotic effects (Figure 1).

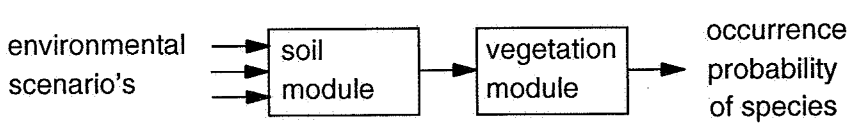

Figure 1 Systematic representation of the multiple stress model for vegetation, MOVE.

In the soil module, changes in abiotic soil factors (soil moisture, soil acidity and nutrient availability) relevant to the occurrence of plant species are predicted. The input consists of acid and nitrogen deposition, and changes in groundwater level as a result of groundwater abstractions. Dynamic soil models (e.g. Simulation Model for Acidification Regional Trends, SMART, De Vries et al., 1989) are used in unfertilized systems. The following processes are included in such models: net uptake and net immobilization of nitrogen; weathering of carbonates, silicates and Al oxides and hydroxides; cation exchange and CO_2 equilibria, and nutrient dynamics. In fertilized systems, the nitrogen load is linked to the vegetation module without dynamic modelling of the soil processes. The nitrogen load in the Netherlands is highly correlated with the plant-species occurrence in fertilized systems (Van Strien, 1991).

The vegetation module predicts the probability of species occurrence as a function of three abiotic soil factors: soil acidity, nutrient availability and soil moisture (Figure 2). With regression statistics the occurrence probability of a species can be calculated for each combination of soil factors or for each environmental variable separately (species-response curve).

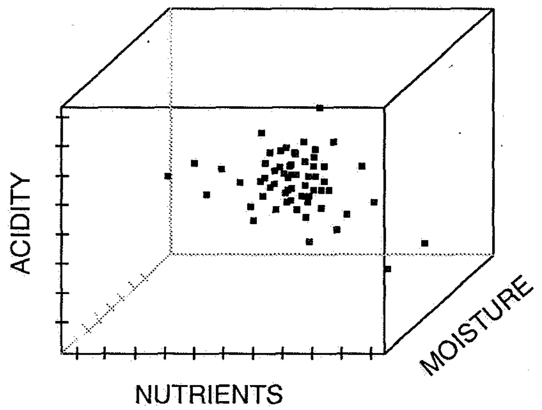

Figure 2 Probability of species occurrence shown in a matrix with dimensions defined by variables related to acidification, eutrophication and desiccation. The occurrence of a certain species in the matrix is indicated with small squares.

Species-response curves of 700 plant species have been described for soil moisture, nutrient availability and soil acidity (Wiertz et al., 1992) using Gaussian logistic regression models. Regression was based on an extensive data base developed for a revision of the Dutch classification of plant communities (Schaminee et al., 1989). This data base consists of 17 000 vegetation relevées. No information on abiotic site factors of the vegetation relevées was available. These were assessed in retrospect with Ellenberg indication values (Ellenberg et al., 1991), using the method of Ter Braak and Gremmen (1987). Ellenberg indication values indicate the relationship between the occurrence of plant species and nutrient availability, acidity, soil moisture, salt dependency, and temperature. These values have been assigned to most plant species of western and central Europe and the Netherlands (Wiertz, 1992). The abiotic site factors of each vegetation relevée are assessed by averaging the indication values of all the species recorded. Calculated averages, in Ellenberg indication values, are used as a semi-quantitative assessment of the abiotic soil factors. Next, the occurrence frequency of each species is established as a function of the averages of vegetation relevées. The occurrence frequency is described with Gaussian logistic regression models (Jongman et al., 1987). Since this analysis uses floristic information to assess the abiotic site factors, any (historical) vegetation sample can be included in the analysis, extending the data base. Moreover, such an analysis excludes potential bias caused by high temporal and spatial variation in the actual measurements of abiotic site factors. Deducing values for the abiotic soil factors from the vegetation sample guarantees ecological relevance.

Species occurrence has been described as being significant for 95% of the species using unimodal and linear regression models. Most of the significant models were unimodel. Linear models were found for nutrients (4%) and salt (20%).

Ellenberg indication values can be calibrated with quantitative values for the abiotic soil factors using combined samples of vegetation and environmental variables. This calibration connects the soil module with the vegetation module. In fertilized grasslands of the Dutch peat area, nitrogen load (kg N $ha^{-1}.a^{-1}$) is, for instance, calculated from Ellenberg indication values using a linear calibration ($p < 0.005$) of the optima for 42 plant species, with quantitative optima for nitrogen load (Latour et al., in press). On the basis of this regression the nitrogen load, N (kg N $ha^{-1}.a^{-1}$), is calculated from Ellenberg indication values (E) with: $N = (E - 2.6) / 0.016$. Calibration for soil acidity, soil nutrients and soil moisture is now in progress for the Netherlands.

We will now go on to describing, with examples, MOVE applications: setting ecological limit values, predicting the ecological effect of scenarios and ranking environmental threats.

3 SETTING ECOLOGICAL LIMIT VALUES

With respect to ecotoxicology, concentrations of various compounds protecting 95% of the species of an ecosystem (VROM, 1991) - the maximum tolerable concentrations (MTC) - have been calculated by extrapolation of single-species toxicity data (e.g. No Observed Effect Concentrations (NOEC)) to an ecosystem. Various extrapolation techniques have been proposed (see Slooff, 1992, for an overview). Similarly, Van der Eerden et al. (1992) calculated critical levels for NH_3 and SO_2 based on NOEC determined in laboratory tests.

Using the multiple stress model for vegetation (MOVE), we have recently adopted the risk assessment for eutrophication and desiccation. The method uses percentiles of the species-response curves as a measure of risk at the species level, analogue to NOEC. For instance, the 5 and 95 percentiles of the species-response curves are used as NOEC-like measures for the risk at the species level (Figure 3). The 5 percentile corresponds with a reduced occurrence probability due to "limitation", the 95 percentile due to "intoxication". Next, the percentage of species protected is plotted for each value of the environmental variable. Species are considered protected if the 5 percentile of the species is lower and the 95 percentile higher than the environmental variable. The 95% protection level of all species can be established from this relationship.

We calculated the potential number of species of grassland in the province of Zuid-Holland as a function of nitrogen, based on species-response curves of 275 plant species from Clausman et al. (1987). The maximum tolerable nitrogen load, corresponding with protection of 95% of the species, is 60 kg N $ha^{-1}.a^{-1}$ (Latour and Reiling, 1992). This is one order of magnitude less than current loads of 450 kg N $ha^{-1}.a^{-1}$ (CBS, unpublished data), resulting in a protection level of about 25% of the plant species.

In the Netherlands ecological targets are currently being specified using a system of 100 so-called nature conservation target types (LNV, 1992). These target types are specified for areas in the Netherlands' ecological network (L&V, 1989). Although still fully in development, the target types are defined as assemblages of plant communities characterized by target species, abiotic conditions and minimum surface area. We are currently assessing ecological limit values of each of these nature conservation target types for soil pH, soil moisture and nutrients, using the risk assessment as described earlier.

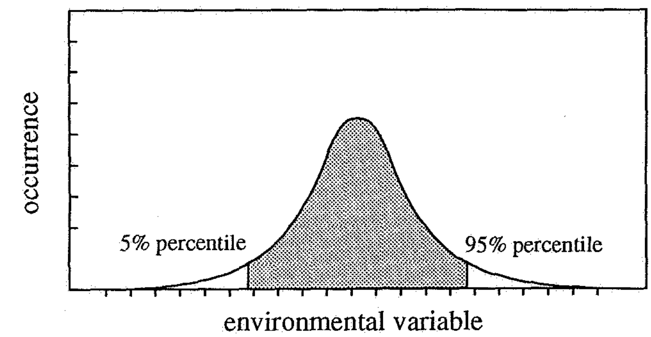

Figure 3 A species-response curve showing the occurrence probability of a species as a function of an environmental variable.

4 CRITICAL LOADS FOR NITROGEN, BASED ON CHANGES IN SPECIES COMPOSITION

The critical-load concept is used as a scientific tool for planning far-reaching and cost-effective emission controls in acidification (Kamari et al., 1992). Critical loads are calculated with mass balance models (e.g. De Vries and Kros, 1991) that describe changes in critical chemical values for ion concentrations or ion ratios concentrations in forest soils, groundwater or tree needles. These critical values are indicative of the ecosystem status (De Vries and Heij, 1991).

The species composition of ecosystems may, however, alter, even though the critical values for ion concentrations and ratios are not exceeded. Bobbink et al. (1992) assessed critical loads for ecosystems based on changes in species composition of flora and fauna. Such changes are derived from various sources of information, such as field and laboratory experiments and comparison of flora and fauna composition in time and space. It is not possible, however, to ascertain if empirical critical nitrogen loads equally protect various ecosystems. Moreover, the current empirical critical nitrogen loads do not incorporate regional differences in, for example, soil buffering capacity and sensitivity of species.

As stated earlier, we used the concept of risk assessment to quantify ecological limit values corresponding to similar protection of ecosystems. Recently, we proposed to calculate critical loads based on changes in species composition systematically and quantitatively using the multiple stress model, MOVE. As an example we calculated critical loads for lowland wet heathland (Latour and Reiling, 1993). Depending on the choice of the desired occurrence probability, critical loads were between 13 and 37 kg N $ha^{-1}.a^{-1}$. Currently, critical loads are being calculated for several ecosystems.

5 PREDICTION

The occurrence probability of a species, groups of species or all species can be predicted for scenarios for acidification, eutrophication and desiccation. As an example, the probability of occurrence for grassland plant species in fertilized grasslands in the Netherlands in 1989 and in 2010 has been assessed as a function of nitrogen load. Ecological response curves of 275 grassland plant species of the province of Zuid-Holland (Clausman et al., 1987) were included in this analysis. It is assumed that response curves in the province of Zuid-Holland do not differ systematically from ecological response curves for other areas in the Netherlands. Species were considered protected if the 95% percentiles of the ecological response curves were not exceeded.

Data on nitrogen load were based on agricultural statistics (LEI-DLO, 1991). The nitrogen load for 2010 was based on the scenario described in Maas (1991). These statistics include average nitrogen loads from manure and fertilizer for each soil type in each region. Location of grasslands was based on remote sensing analysis of land use. The percentage of species for which the 95% percentiles did not exceed is calculated for each region. Results are expressed as potential diversity. In most regions situated on sandy soils of the central, eastern and southern parts of the Netherlands, the potential diversity is less than 10% (Figure 4). In most regions situated on peat and clay soils of the western and northern parts of the Netherlands, the potential diversity varies between 10% and 30%. In some regions potential diversity is higher, namely up to 60%. If policy objectives for reduction of nitrogen loads are implemented (scenario in Maas, 1991), the potential diversity will increase by an average of 10% (Figure 5).

Locally, potential diversity may deviate from regional averages since obviously the nitrogen load will vary within regions. Data on the variance of nitrogen load within regions and individual farms is not currently available. Consequently, the actual diversity will deviate from potential diversity. Actual diversity can be mapped using extensive floristic data bases (Groen et al., 1992).

Currently, a feasibility study on using the MOVE model for multiple stress modelling on a European scale is being conducted. For a limited number of species the probability of occurrence in Europe is determined from the combined effect of global (climatic change), continental (acidification) and regional (desiccation and eutrophication) environmental problems.

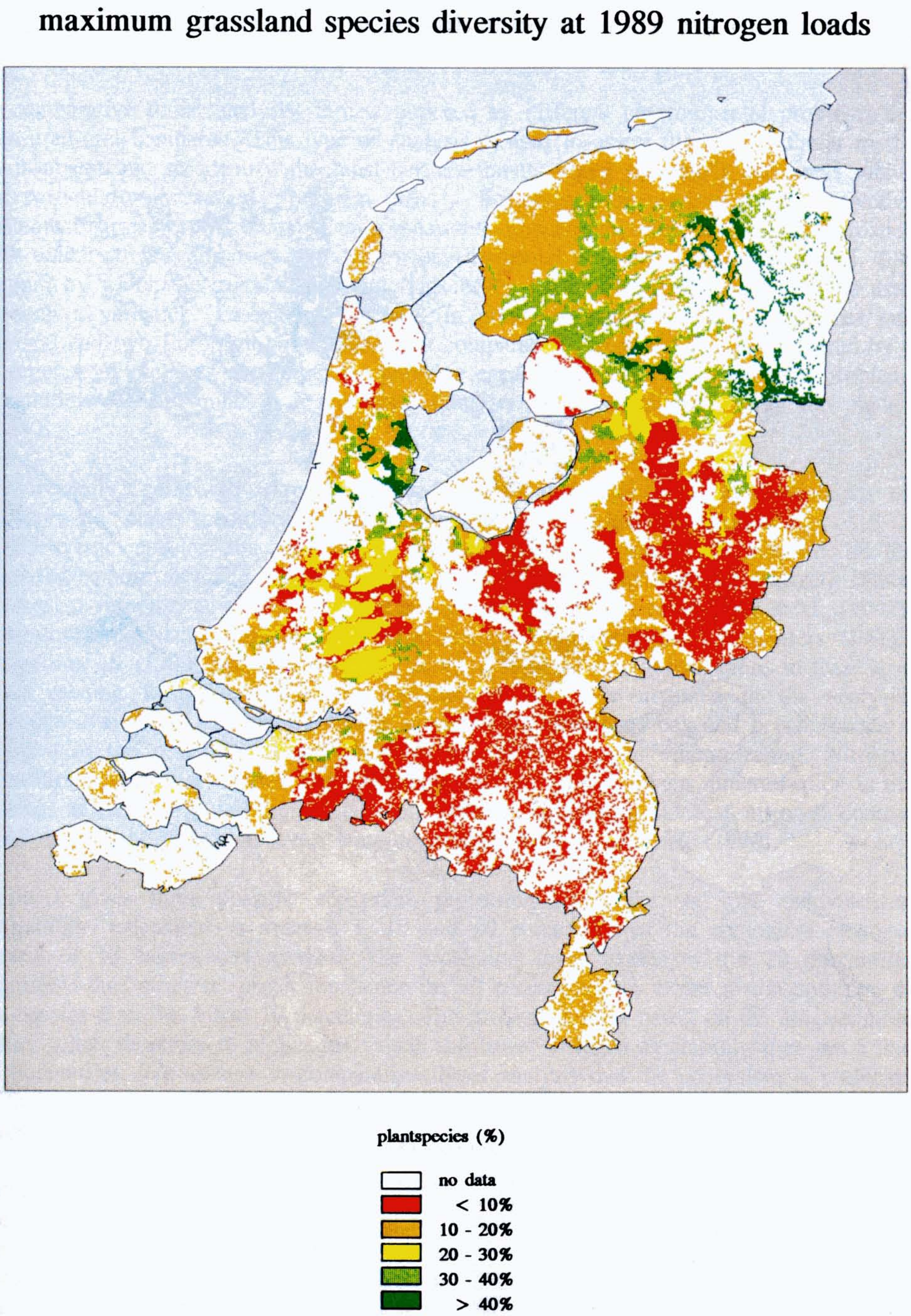

Figure 4 The potential diversity of grasslands in the Netherlands on fertilizer and manure loads based on a set of 275 ecological response curves for plant species in the province of Zuid-Holland. It is assumed that response curves in the province of Zuid-Holland do not differ systematically from ecological response curves in other areas in the Netherlands.

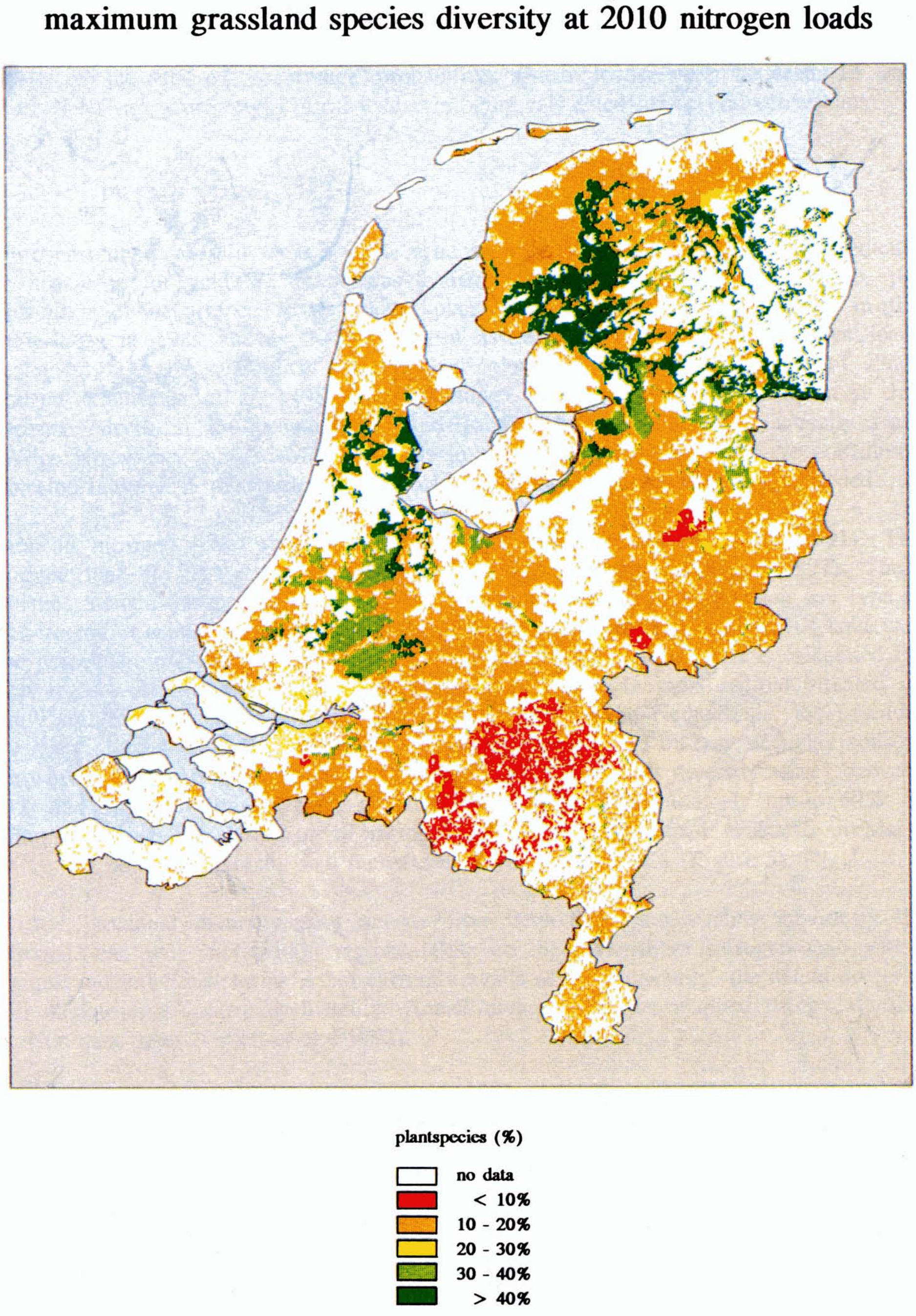

Figure 5 As Figure 4 but for the year 2010 using a scenario in Maas (1991).

6 RANKING ENVIRONMENTAL THREATS TO SET ABATEMENT PRIORITIES

In comparative threat analysis, threats imposed by different environmental problems are measured and compared. This type of analysis is used to assess the overall threat to the environment and to identify the most serious threats ("key problem") and areas which are particularly threatened ("problem areas"). Results can be used to set cost-effective abatement strategies. Elementary to comparative threat analysis is a common yardstick with which stresses imposed by different environmental problems can be compared. The amount by which the carrying capacity is exceeded by environmental loads may be used as such a yardstick. Latour and Reiling (in press, b) conclude on the basis of one national and two European case studies that comparative threat analysis is hampered by a divergence of methods describing the carrying capacity for each environmental problem. A solution can be found in a risk-based comparative threat analysis using the model, MOVE.

We recently compared threats imposed by acidification and eutrophication on undergrowth plant species of nutrient-poor deciduous forest (Latour et al., 1993). Characteristic species for nutrient-poor acidic deciduous forest are implied in the *Quercion robori petraeae* and *Fago quercetum* (Loopstra and Van der Maarel, 1984). Ecological response curves of 13 plant species were derived from Wiertz et al. (1992). Data concerning actual pH of soils were derived from De Vries and Leeters (1993), Bongers et al. (1989) and Van den Berg et al. (1990). Sixty-eight locations in these data bases relating to nutrient-poor dry deciduous forest were included in the analysis. Nitrogen availability of plant species in nutrient-poor deciduous oak and beech forests is assessed as the sum of nitrogen deposition and gross nitrogen mineralization. Nitrogen deposition levels were derived from Erisman (1991). Gross nitrogen mineralization is the product of leaf mass and leaf N content. Data on leaf mass and leaf nitrogen content under various N deposition levels were derived from De Vries et al. (1990).

Table 1 gives three possible ecosystem protection levels for soil pH and nitrogen availability, calculated as average 5, 10 and 20 percentiles of the ecological response curves of 13 herbaceous species. The protection level based on the 20 percentiles represents the "strictest" protection, since the 20 percentiles are closer to the optimum of the species than the 5 and 10 percentiles. The protection level based on the 20 percentile values refers therefore to higher pH values and lower nitrogen availability than the 5 and 10 percentiles. We assume in comparative threat analysis that the ecosystem is protected if the nitrogen availability load is lower and the soil pH higher than the protection levels.

Table 2 shows the percentage of locations in which the protection levels are exceeded because of nitrogen availability and soil pH. There are more locations in which species are at risk as a result of nitrogen availability rather than as a result of soil pH. Differences are significant for the 5 and 10 percentile levels, as indicated by the confidence intervals (Table 2). Excessive nitrogen availability (eutrophication) thus represents a more serious threat to plant species of deciduous forests growing on nutrient-poor dry acidic sandy soils than soil pH (acidification).

Table 1 Ecosystem protection levels for soil pH and nitrogen availability for deciduous forests on nutrient-poor dry acidic sandy soils, calculated as the average 5, 10 and 20 percentiles of the ecological response curves of 13 herbaceous species (Latour et al., 1993).

Protection levels	Soil pH	Nitrogen availability ($kg\ N\ ha^{-1}.a^{-1}$)
Optimum	4.26	46
20 percentile	3.84	76
10 percentile	3.61	91
5 percentile	3.43	104

Table 2 Comparison of acidification risks (soil pH) and eutrophication (N availibility) for deciduous forests on nutrient-poor dry acidic soils (Latour et al., 1993). Shown are the percentages of forest stands (n=68) is given in which the optimum, 20, 10 and 5 "percentile" protection levels are exceeded because of soil pH and nitrogen availability, respectively. Confidence intervals are given in parentheses.

	Measured soil PH (%)	Modelled nitrogen availibility (%)
Optimimum	91 (82-97)	100 (95-100)
20 percentile	75 (63-85)	82 (71-90)
10 percentile	38 (27-50)	71 (58-81)
5 percentile	15 (7-25)	51 (39-64)

7 DISCUSSION

We reviewed a species-centred modelling approach that can be used for prediction, setting ecological limit values and ranking environmental problems.

The functional soil model, SMART, used in the MOVE soil module, will be linked with the probabilistic species-centred vegetation module. The latter is based on an extensive data base, which may be extended and optimized in the future. Both abiotic and biotic modules may be used to indicate ecosystem stress. Changes in nutrient cycling and increased leaching of nitrogen, as modelled in the soil module, are often used as ecosystem health indicators (e.g. Rapport, 1989; Ulrich and Bredemeijer, 1990). Changes in species composition as modelled in the vegetation module may provide an early warning of ecosystem collapse (Rapport et al., 1985). Changes in species composition are also relevant if species are target species (V&W, 1989; LNV, 1990).

MOVE predicts whether or not the conditions conducive to species occurrence are present as a function of (general) environmental policy. Whether or not a species actually occurs at a given location, evidently depends on many environmental factors not incorporated in the model. For instance, the model does not take local differences in management practices into consideration.

Several constraining factors are responsible for the shape of the ecological response curve. Considering an ecological response curve for nutrients, the "nutrient-poor" side of the ecological amplitude may reflect physiological constraints of a species' nutrient metabolism in combination with reduced competitive species abilities. The "nutrient-rich" side may be dictated by nutrient intoxication, also in combination with competition. We consider such competition as an integral part of the nutrient stress in real-world circumstances.

The vegetation model MOVE strongly resembles the regional species-centred model ICHORS, as described by Barendregt and Nieuwenhuis, 1993 (this volume). This model predicts the occurrence probability of each individual species as a function of 22 chemical and physical variables of soil, groundwater and surface water, using samples in which abiotic site factors and vegetation have been sampled simultaneously. It is, however, unlikely that these models can be developed for the whole of the Netherlands in the near future, because extensive field inventories are needed as input for the model's regression analysis. Moreover, these regional models do not simulate changes in abiotic site factors, as in the abiotic soil module of MOVE. Instead, they use predefined values for the abiotic site factors as input data. These regional models may, however, be used for specific regions for which they are available to improve the relatively more general vegetation module of MOVE.

We did not use ecosystem classification systems for prediction of multiple stress. Such systems have been used in several regional and national studies, particularly on the topic of desiccation (e.g. Witte et al., as described elsewhere in this volume). However, classified ecosystems generally do not respond as coherent entities to changing conditions. The response entity is, in principle, the species, all of which respond differently to large environmental changes. Non-analogue ecosystems could emerge in the dynamic response processes. In modelling ecosystem response to climatic changes, rigid ecosystem clasifications have also been criticized (Leemans, in press). Leemans therefore uses models based either on species characteristics defined by their physiological characteristics or, if not possible, on functional types.

The quality of the regression analysis depends on the data base used for the analysis. Wiertz et al. (1992) point out that the data base of Schamminee contains a reasonable number of relevées of almost every plant community (>20 per alliance) and shows an equal distribution of the site factor values over almost all the classes. However, as botanists have a preference for sampling on ecologically interesting sites, the data base will contain relatively few relevées of less interesting sites. In the future we will test to what extent this influences the results and to what extent it can be corrected.

We believe that MOVE can be extended to other European countries. The soil model, SMART, was developed on a European scale in the context of critical-load studies. The Ellenberg indication values, needed in the vegetation module, have been assigned for most European species. A prerequisite for an analysis of MOVE in other countries, however, is a digital data base of vegetation relevées with which ecological response curves can be calculated, and a set of relevées with abiotic measurements for the calibration of the Ellenberg values. Auxiliary data include soil maps, deposition data and flora atlases.

Currently, the calibration of Ellenberg indication values in abiotic soil factors is being quantified, and the model SMART adjusted. Moreover, the procedure of standard-setting is currently under study. Underpinning selection of percentiles and the species protection level will be pursued and the results compared with the known ecological limit values of a limited number of well-researched plant communities. In this chapter the primary intention of using the 95% protection levels of all species and the 5 and 95 percentiles of species-response curves is to illustrate the method. The results of these are therefore only tentative.

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LINKING HYDROLOGICAL AND ECOLOGICAL MODELS

A.C. Garritsen

ABSTRACT

The hydro-ecological models DEMNAT, ICHORS and MOVE were compared using the characteristics in Table 1 and a subdivision into five levels of the hydrological-ecological cause-effect chain.

Level 1 is the water management system in a region. Modifications on that level lead to changes on level 2: the regional hydrological system. Level 3 is the local hydrological system of a nature reserve or a comparable landscape unit. Within such a nature reserve several sites may be distinguished: level 4. The vegetation is level 5. The full prediction sequence covers four steps between a modification of the water management system and the effect on the vegetation.

The application of the models is restricted by three types of problems:
1. availability of adequate models;
2. connection between the models for the different steps;
3. reliability of the single model steps and error transmission.

1 INTRODUCTION

The Netherlands is a country with large surface waters, fresh as well as salty, and a shallow groundwater table in major parts of the country. About 25% of the land area lies below Mean Sea Level and in absence of dunes and dikes 65% of the country would be flooded at high sea and river levels. The lower part of the country and the former river flood plains consist entirely of polders, where water levels are controlled by pumping excess water into the storage canals. In summer the direction of water movement may be reversed to prevent a fall in water level. Since almost the entire country lies in the delta areas of three large rivers, the Rhine, Meuse and Scheldt, ample supplies of fresh surface water are available in large parts of the country (Colenbrander, 1986).

This particular situation has brought about a wide range of both necessities and possibilities concerning water management. The most important is the need to protect the land from flooding. Water levels have to be controlled in areas of agriculture, navigation, foundations of buildings, industrial and household water use, recreation and - finally - nature. This combination of needs and wishes has led to a complex system of water management in the Netherlands. The water management policy in the past few decades was primarily dedicated to agricultural needs and drinking-water supply. This has caused a major environmental problem which has only recently been acknowledged: **"verdroging"**.

"Verdroging" may be defined as the ecological decline of nature reserves and natural elements in the agricultural landscape due to side-effects of human activities that affect the hydrologic cycle, such as drainage of wetlands and withdrawal of groundwater (Projectteam Verdroging, 1989). Translation of this term is as difficult as translating the word **"polder"**. Both words are strongly related to the typical Dutch water management system.

Three mechanisms are considered to be the main hydrological cause of **verdroging.** These are:
1. a direct draw-down of the groundwater table;
2. the decrease or total disappearance of upward seepage;
3. the supply of a region with river water to prevent or compensate an artificial draw-down of the groundwater level.

Lowering of the groundwater level is observed in large parts of the country. The most frequently observed magnitude of lowering is 30 cm. The lowering of the groundwater level is often found to be greater in deep wells than in shallow ones (Garritsen et al., 1990).

More than 50% of the area of groundwater dependent vegetations in nature reserves and semi-natural parts of cultivated land is affected by **verdroging**, especially in the higher parts of the Netherlands. Most affected are vegetations on oligotrophic and mesotrophic sites in the coastal dunes, peat bogs, moist heathlands, marshland forests and meadows (Van Gool et al., 1990).

Due to this ecological decline people concerned with the management of nature reserves realized that water, and consequently also the management of water, is of major importance to nature management and conservation. With the acknowledgement of **verdroging**, nature conservation and restoration became a new aspect in water management policy. These developments have caused an increasing effort of Dutch scientists to study the relation between water, mainly groundwater, and spontaneous vegetation. This led to the development of ecohydrology: the interdisciplinary field of research directed to the application of hydrological knowledge in landscape ecology (Pedroli, 1990). Both ecologists and hydrologists are working in this field of science, which is aimed at applications in water management and water policy. Several methods and models in hydro-ecology and ecohydrology have been compared before (Wassen en Schot, 1992; Van Wirdum, 1985).

In this paper three models are compared in detail: DEMNAT, MOVE and ICHORS. For this comparison and for the distinction between hydro-ecological and ecohydrological modelling a subdivision of the cause-effect chain in five levels is presented first.

Furthermore, a summary of limiting factors in the current modelling is presented. In the concluding part possible future developments are discussed.

2 FIVE LEVELS IN THE CAUSE-EFFECT CHAIN

2.1 Introduction

Hydro-ecological and ecohydrological models and methods are used to predict the consequences of a change in water management on one or several ecosystems. Normally the prediction is carried out for a limited geographic region. Furthermore, in describing the effects, the ecosystem is generally reduced to the vegetation. This is done for three reasons: firstly, because of the important role of the vegetation in the ecosystem; secondly, because the vegetation is the most directly influenced part of the ecosystem and finally, because vegetation is the most simple component to study.

In order to compare these models and to distinguish between ecohydrology and hydro-ecology, a subdivision in the cause-efffect chain in five levels is presented, which is shown in Figure 1. Each level is associated with certain types of data and an appropriate degree of detail which is required for further use.

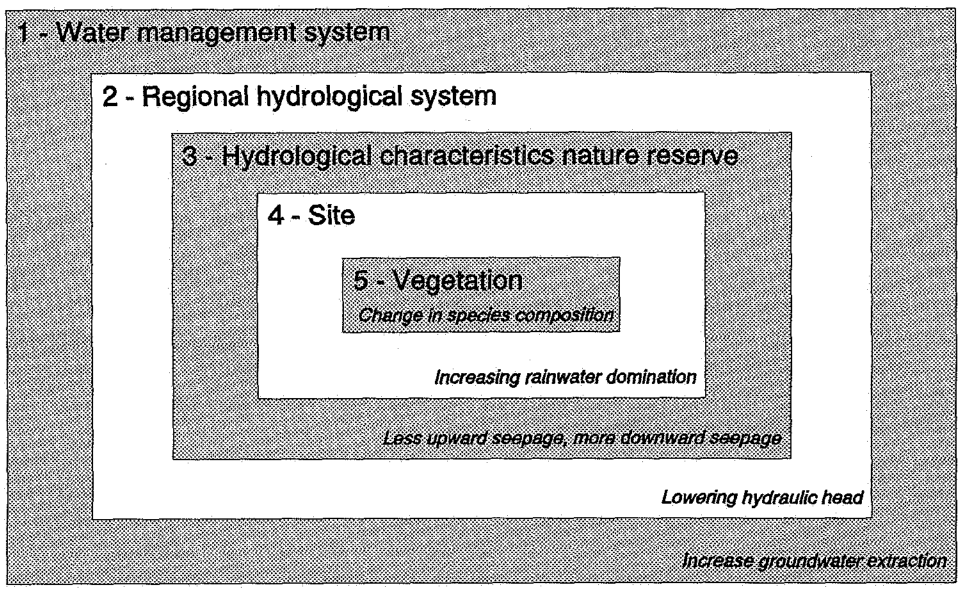

Figure 1 Subdivision of the hydrological-ecological cause-effect chain into five levels.

Level 1 comprises the water management system of a region. Modifications in the water management on this level lead to changes on level 2: the regional hydrological system. Level 3 comprises the hydrological situation and characteristics of a local nature reserve or comparable landscape element. Within such an element many sites may be distinguished. A site may be defined as a small geographic unit which is homogeneous regarding the abiotic factors which control the response of spontaneous vegetation (Stevers et al., 1987). The site itself is situated at level 4, the vegetation finally at level 5.

2.2 Example

For example: a groundwater extraction well field in the undulating coversand area of the province of Noord-Brabant, may be described using parameters concerning the discharge in million cubic metres per year, the number and location of wells and the depth of the filters (level 1).

As a result of this groundwater extraction the regional hydrological system changes. First there is a lowering of the hydraulic head of the groundwater in the aquifer from which the water is extracted. This lowering spreads to other aquifers and the phreatic surface. This changes the flow direction of the groundwater and as a consequence also the groundwater composition, though the latter change takes place slowly. These changes belong to level 2.

The nature reserve in a brook valley is part of this regional hydrological system. Nevertheless it is distinguished as a separate level, level 3, because the hydrological changes on this level must be predicted and described in a different way than at level 2. More details are needed in space, for example by using a grid with elements of 10x10 metres, and in time, for example by predicting a graph of groundwater level versus time instead of one annual or two (summer and winter) average values. Relevant parameters on this level are not only the hydraulic head and the phreatic surface but also the direction of flow (primarily the condition of upward or downward seepage) and the quality of the groundwater.

In this nature reserve several sites (level 4) are present. As a result of the changes on higher levels, a site on the flank of the valley with a sandy soil may become dryer, whereas a site in the lower part of the reserve near a ditch may shift to an area more dominated by rainwater, acid environment whilst remaining as moist as before. In the centre of a strip of land between two ditches again other changes will take place. Important parameters on this level are the depth and fluctuation of the phreatic groundwater level, the composition of soil moisture and shallow groundwater and the structure and composition of the soil.

Finally the vegetation is affected. The moist heath (*Ericetum tetralicis*) on the flank will change in dry heath with, for example, the disappearance of *Erica tetralix*. The aspect of the *Calthion palustris* vegetation of the site in the lower part will shift in the direction of a more acid abiding vegetation. These descriptions of level 5 concern the absence and presence and sometimes also the abundance of plant species and syntaxonomic associations.

2.3 Linking the levels

In modelling the ecological effect from water management modifications, four steps have to be taken. These steps, which link the levels 1 through 5, are indicated in Figure 2 with the letters A, B, C and D.

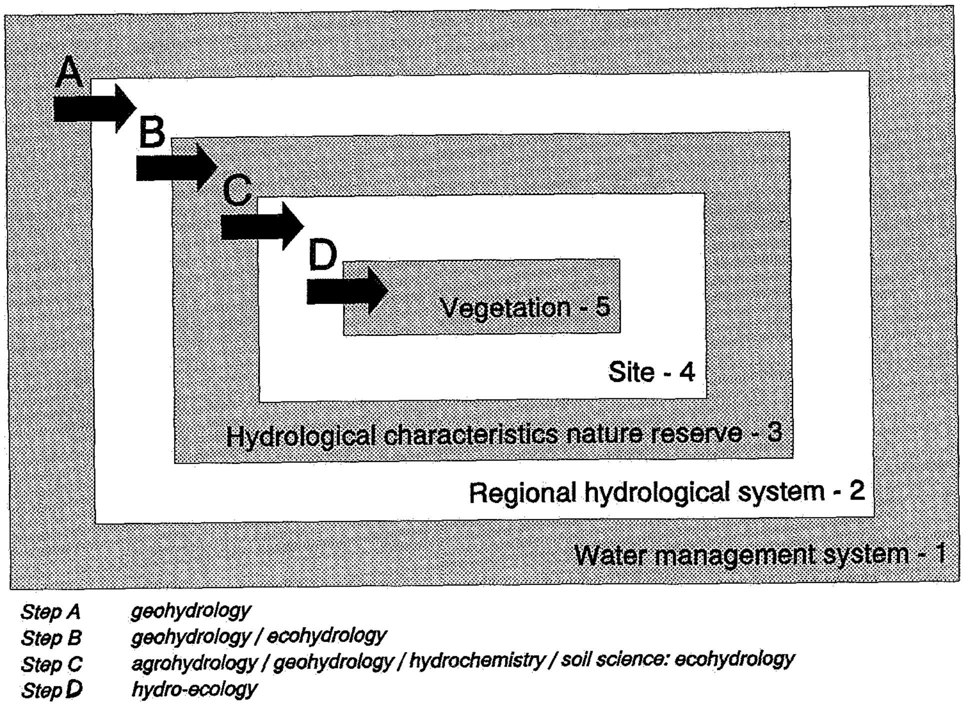

Figure 2 The steps A, B, C and D in hydrological-ecological modelling.

The first step (A) links the water management modification to a change in regional hydrology. A hydrological model is needed. Groundwater often is of primary concern, therefore a geohydrological model is mostly used. In the next step (B) such a model is possibly adapted and in any way applied with special attention to the detail which is needed for the prediction of changes on the level of nature reserves. This step could be called ecohydrological modelling since it comprises primarily hydrological modelling to supply data for ecological modelling. For the third step (C) input from several disciplines is needed to answer the question how the change in hydrological characteristics of a nature reserve affect the sites. In order to do so various adequate modelling is needed of the functioning of ditches and trenches and the hydraulic resistance and storage in the soil and subsoil. The same applies to the soil chemistry in relation to water. The final step (D) links the abiotic characteristics of the site to vegetation and may therefore be called hydro-ecological. Whereas the first three steps involve spatial modelling in two or three dimensions, this step is actually non-dimensional.

It should be remarked that the four mentioned steps are not the only relations that link the five levels. Changes in vegetation are not necessarily caused by a change in the abiotic site conditions. A clear example is the importance of vegetation management such as cutting or cattle grazing. The site itself can also be influenced by other than hydrological factors, for example by atmospheric deposition. Furthermore, not all the hydrological factors which control the site characteristics are dependant on the level 1 water management system. An important factor is the internal hydrological management in the nature reserve. Examples of such management practices are the damming or damping of ditches and the making of trenches. Under certain circumstances this internal water management may influence the level of the nature reserve as well as the site level.

3 A COMPARISON OF THREE MODELS

3.1 Comparison

In Table 1 a comparison is given of the most important characteristics of the models DEMNAT (Witte et al. 1992; Witte et al., 1993, this volume), MOVE (Latour and Reiling, 1992; Latour et al., 1993; Latour et al., 1993, this volume) and ICHORS (Barendregt and Wassen, 1989; Barendregt and Nieuwenhuis, 1993, this volume). The models DEMNAT and ICHORS are frequently applied in the Netherlands, MOVE is partly still under development.

Considering the type of input, MOVE and DEMNAT are comparable as far as hydrological events are involved.

In the handling of site characteristics, the three models are totally different. MOVE is designed to operate with a process-oriented soil model (De Vries et al., 1989) and DEMNAT combines several types of geographical information to determine ecoplots. ICHORS does not include a module to predict changes in abiotic site characteristics since it uses descriptions of abiotic site conditions as input.

For the hydro-ecological step (step D) in the cause-effect chain, plant response functions are required, which are present in all of the three models. DEMNAT is totally different from MOVE and ICHORS since it uses relations for specific ecosystem types (*ecotopes*) as an ecological unit, whereas MOVE and ICHORS use plant species. DEMNAT uses relations for moisture, pH and nutrients. MOVE combines the Ellenberg-indication numbers (Ellenberg, 1986) for moisture, nutrients, pH and salt, with a large number of field observations of plant species using single regression statistics to predict the probability of occurrence of plant species (Wiertz et al., 1992). ICHORS uses step by step logistic (multiple) regression to determine correlations in a data base of field observations containing a large number of abiotic site variables on one hand and the occurrence of plant species on the other hand.

Consequently the type of output of MOVE and ICHORS are comparable. Both models produce probabilities of occurrence of species. The spatial detail of this output, however, is different. MOVE is used to predict probabilities per geographical unit of one square kilometre. ICHORS itself is dimensionless, so the scale of the output is related to the scale of the input and may be as detailed as required up to predictions for units of just a few square metres.

DEMNAT is quite different since it predicts estimates of the completeness of ecosystem types and estimates of nature value. DEMNAT is used in predictions for geographical units of one square kilometre.

Table 1 Characteristics of three hydro-ecological models

	DEMNAT	MOVE	ICHORS
type of input	changes in the regional hydrological system	changes in the regional hydrological system changes in deposition	description of non biotic site conditions
handling of site characteristics	overlay of soil and phreatic level maps to determine ecoplots	process-oriented soil model SMART - under development	not included in the model
plant response functions	field observations and relations from literature ------> relations for moisture, pH and nutrients	Ellenberg-indication numbers and 16.000 field observations & regression statistics ------> ecological amplitudes for moisture, pH, nutrients and salt	field observations on species and non biotic parameters & regression statistics ------> multiple optimum curves for many parameters
	ecosystem types	*species*	*species*
type and spatial detail of output	km^2	km^2	dimensionless
	completeness of ecotopes, estimate of nature value at specified grid map units	*probabilities of occurrence for species (species may be grouped) at specified grid map units*	*probabilities of occurrence for species (species may be grouped) at specific locations; to be extrapolated to areas*
applicability type of region	the Netherlands excluding brackisch and salty parts	the Netherlands, parts of Europe	Holocene parts of the Netherlands, polder areas
	terrestrial and aquatic vegetation	*terrestrial vegetation*	*aquatic and semi-terrestrial vegetation*
applicability type of problem	policy	policy	policy and management
	national and regional	*national and regional*	*regional and local*

The applicability of the models is discussed considering two arguments: (1) type of region and (2) type of problem. Concerning the type of region ICHORS is the most limited of the three models, since it has been applied only to Holocene parts of the Netherlands. Furthermore, ICHORS is limited to aquatic vegetation, the vegetation of banks and the vegetation of wetlands. MOVE has been applied to the situation of the Netherlands as a whole and tentatively to parts of Europe as well, but it is limited to terrestrial vegetation. DEMNAT produces predictions on terrestrial as well as aquatic vegetation and may be applied to all but the brackisch and salty parts of the Netherlands.

As a consequence of the regional characteristics and the spatial detail in the output, DEMNAT and MOVE may be used for problems concerning policy on a national and regional level, whereas ICHORS is suitable for both policy and management problems on a regional and local level.

3.2 Cause-effect chain

The three models under consideration cover different steps and levels in the cause-effect chain, as shown in Figure 3.

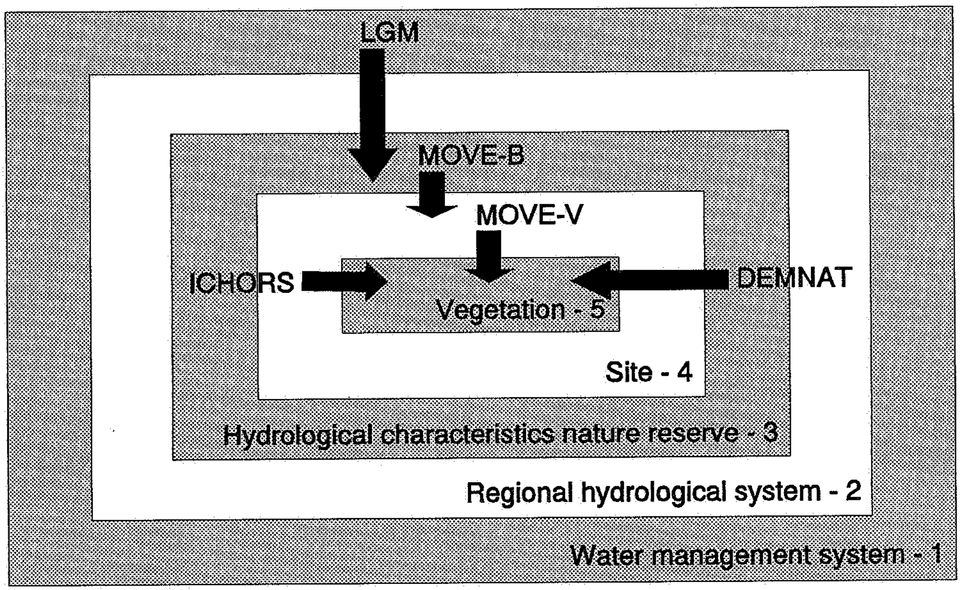

Figure 3 ICHORS, MOVE and DEMNAT positioned in the cause-effect chain.
MOVE-V = vegetation module
MOVE-B = soil module

ICHORS covers only step D from level 4 to 5. In the application of this model several other models have been used to cover the other steps (Van Pruissen and Hemker, 1988; Grootjans et al., 1990).

MOVE covers the steps C and D with respectively the soil module SMART and the vegetation module. Prior to the application of MOVE the Dutch National Groundwater Model (abbreviation in Dutch: LGM; Pastoors, 1992) is to be used.

DEMNAT tackles the steps C and D in combination. Prior to DEMNAT two models have to be used: DEMGEN (surface water distribution and quality) and NAGROM or LGM, both are national groundwater models (Witte et al., 1992).

4 RESTRICTIONS

The accuracy of the presented and similar models is restricted by several factors which may be divided into three categories.

The first category concerns the single steps in the sequence of Figure 2. For several steps adequate instruments are not available, or, if they are available, they are generally not used in the application of hydro-ecological models.
Examples of the latter are:
- the modelling of the unsaturated zone and the so called geohydrological "top-system" (hydraulic resistance and the functioning of ditches, drains and trenches) concerning water quantity on level 4 (step C);
- the modelling of surface water levels and flow up to the presence of such water in small trenches in connection with the groundwater situation (step C).

Examples of the absence of adequate models are:
- the prediction of groundwater quality on several levels (steps A, B and C);
- the role of phosphate as a site condition for the vegetation (step D).

The second category concerns the cumbersome connection between the different steps, especially between B, C and D. Mostly, details in space and time are lost or cannot be taken into account (Van Bakel, 1992). Apart from these, other scale problems may occur, for example, when the scale of the model results and the scale of the questions to be answered are very different (Hooghart, (ed.), 1993). For this reason, DEMNAT and MOVE are not used for local problems.

The third category concerns the reliability of the single model steps and error transmission through the entire sequence. The reliability of the single model steps is mostly unknown. Models like ICHORS and MOVE show probabilities of the occurrence of species, but this probability is exclusively a result of the applied statistical method and the variation in used data. Furthermore, more research has to be done on the sensitivity of the various models for errors in input that was predicted using another model to cover the previous step. For example, if the model for step B produces a result with a possible error of 20% and the model for step C turns out to be very sensitive for errors in this input data, the error may become much larger than the initial 20%. As a result of this adding and multiplying of errors, the final prediction of the entire sequence may have an unacceptably large potential error. Considering these lacks of knowledge it is a severe shortcoming that a true validation of the models under consideration has never taken place (or at least never has been published). Here, only a comparison of results from an application of the model with reality is considered a true validation. In practical terms this would need the following: measure and describe the initial situation, describe an intended modification in regional water management, predict effects using the entire model sequence, perform the modification, monitor the new situation in the course of up to more than 10 years after the modification in the water management system and, finally, compare the results.

Considering these bottle-necks, especially the error transmission and the lack of true validations, one may wonder whether these models are adequate for solving the problems they are currently used for.

5 DISCUSSION ON CURRENT USE AND FUTURE DEVELOPMENTS

The current use and future developments of the models is discussed using two questions as a starting point. Some opinions of the discussion panel members at the meeting of the TNO Committee on Hydrological Research (CHO-TNO) held in Ede, the Netherlands, on 25 may 1993, may be recognised. However, this paragraph is not a summary of that discussion.

5.1 Are the models good enough?

Are the presently available models good enough for the current use in policy and nature management? In discussing this question the following three aspects are considered: applicability of the model, applicability of the model output and reliability of the output.

In spite of the fact that the models have a limited proven validity, the use in policy problems is satisfactory. For policy problems a decision must be based on a limited comparison of alternatives and effects. A global indication of the expected effects is sufficient. Detailed descriptions of effects are not necessary, even unwanted. Excessive information is a burden for policy makers and has a scientifically questionable significance. Models for these types of problems are to be considered primarily as strategic instruments.

For nature management problems, however, the models are generally not good enough. They are difficult to be used, have a limited availability and the applicability of the output (scale and size) as well as the reliability are not sufficient. Furthermore, ecologically valuable, undisturbed situations are generally rare and are consequently not implemented well enough. The same applies for problems involving hydrological restoration and nature rehabilitation.

5.2 How can we make better models?

The currently available models are scientifically still unsatisfactory and not good enough for handling management problems. Consequently, there is a need for better models. The question arises whether such models can be developed and, if so, how this should be done.

For the steps A, B and C (geohydrology and ecohydrology) the physics of water flow are at hand. This is an important difference from the step D (hydro-ecology) where underlying processes are not always sufficiently known. Relations for the step D models are often derived with regression statistics. The possibilities for extrapolation of a local model with a local dataset are very limited, especially for models with statistically derived correlations.

The steps C and D are to be improved using deterministic models. This may initially be possible only when a large amount of data is available. Research on vegetation patterns as well as research on ecohydrological (step C) and hydro-ecological (step D) processes may yield extra data. Furthermore, the key factors have to be determined to build models with limited complexity and to limit the necessary amount of field work.

Both the development as well as the application of these models will require large amounts of data. To facilitate the access to relevant data it is recommended to make a common data base for ecohydrological and hydro-ecological data and to intensify long term monitoring of soil, water and plants in nature reserves and possible nature restoration areas.

It may be impossible for either of these steps to find a set of key factors that may be measured with acceptable effort. In that case, nature turns out to be too complex to be predictable; extrapolation will not be possible and modelling will be too difficult for general application. Obviously a trade-off exists between effort in modelling and detail in predictions. Only more multi-disciplinary research and validation of present and future models will supply answers to questions on the balance in this trade-off and on the predictability of nature.

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