NN08201, 2626

1. Al is een model nog zo snel, vragen van beheerders achterhalen haar wel (dit proefschrift).

2. Als gevolg van voortgaande ontwikkelingen in hard- en software worden landelijke waterkwaliteitsmodellen steeds meer verfijnd, waardoor regionale waterbeheerders in toenemende mate kunnen profiteren van landelijke modelstudies.

3. Informele en persoonlijke relaties en toevallige factoren spelen, naast prijs en kwaliteit, een belangrijke rol bij de keuze van een waterbeheerder voor (de uitvoerder van) een model (dit proefschrift).

4. Betrouwbaarheidsintervallen rond uitkomsten van modellen suggereren meer over zekerheid dan vaak wordt aangenomen (dit proefschrift).

5. Gedetailleerd experimenteel onderzoek naar de verschillende interacties binnen een enkel watersysteem levert meer kennis op over het functioneren van ecosystemen dan spreiding van dit onderzoek over meerdere systemen (dit proefschrift).

6. Zonder actief biologisch beheer waren uitgebreide velden met kranswieren ook wel ontstaan in het Wolderwijd *(Bijlagen D en E van dit proefschrift)*.

7. Bij het uitkomen van de vierde Nota waterhuishouding bleek de credibility van de modelresultaten totstandgekomen in het project Watersysteemverkenningen, onderdeel van de wetenschappelijke verantwoording van de Nota, acceptabel te zijn.

8. Het fanatisme van sommige ecologen met betrekking tot de instandhouding van oorspronkelijke ecologische gemeenschappen vertoont overeenkomst met de ongerustheid van sommige autochtonen met betrekking tot het bewaren van eigen cultuur.

9. Natuur is wat je tegenkomt als je van de kroeg in Rotterdam naar een kroeg in Den Haag rijdt (Jules Deelder).

10. Geslotenheid gedurende de totstandkoming van een proefschrift gaat ten koste van de acceptatie en het gebruik van de resultaten er van.

11. Het gegeven dat pas ruim honderd jaar na het bedenken van de Booleaanse algebra de daarop gebaseerde computer-technologie veel in de samenleving heeft veranderd, illustreert dat fundamentele wetenschap niet altijd direct tot bruikbare resultaten hoeft te leiden (George Boole, 1854, An investigation of the laws of thought, on which are founded the mathematical theories of logic and probabilities; London, Macmillan).

12. Het millenniumprobleem is een gevolg van kortzichtigheid.

13. Het is tijd voor een nieuw volkslied.

14. Sinterklaas bestaat.

Stellingen bij het proefschrift: 'The role of eutrophication models in water management'

Diederik van der Molen
Wageningen, 9 juni 1999

The role of eutrophication models in water management

Diederik van der Molen

aan mijn moeder

ter nagedachtenis aan mijn vader

Promotor: dr. L. Lijklema
hoogleraar in het Waterkwaliteitsbeheer
Co-promotoren: dr. ir. J. Leentvaar
hoogleraar Integraal Waterbeheer
dr. P.C.M. Boers
senior projectleider Water Systemen Ecologie van het Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling te Lelystad

The role of eutrophication models in water management

Diederik van der Molen

Proefschrift
ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van de Landbouwuniversiteit Wageningen,
dr. C.M. Karssen,
in het openbaar te verdedigen
op woensdag 9 juni 1999
om half twee des namiddags in de Aula.

Ministry of Transport, Public Works and Water Management
Institute for Inland Water Management and Waste Water Treatment (RIZA)
P.O. Box 17, 8200 AA Lelystad, The Netherlands
RIZA report 99.020
ISBN 9058080439

Drukkerij Ponsen & Looyen BV Drukkerij
Cover by Tjitse and Yke van der Molen
Lay-out by Grafisch Atelier Wageningen, Harry van Oosterhout

BIBLIOTHEEK
LANDBOUWUNIVERSITEIT
WAGENINGEN

Contents

Abstract

This thesis deals with the credibility and the acceptability of eutrophication models. Credibility is defined as the technical appropriateness of the model and its results. The assessment of credibility is generally a task of the modeller and comprises that the procedure of systems analysis is followed and that uncertainties involved with predictions are considered. Acceptability is defined as the managers' perception of the practical value of the model or its results. Credibility and acceptability are specified in a set of criteria. These criteria are applied to the reviewing of previous work of the author, presented in the Appendices, and for evaluating a number of projects on modelling eutrophication. The results may be useful as guidance for further model development and for model application in water management in The Netherlands.

De sjimpansee doet niet mee
Waarom doet de sjimpansee niet mee
De sjimpansee
is
ziek van de zee
Er gaat zoveel water in de zee
Meent de sjimpansee

Paul van Ostaijen

1 Introduction

Models are nowadays indispensable tools to gain a better understanding of the complexity of eutrophication processes and to guide water managers in making decisions. Specifically, simplifying the complex, real world into models may help

- to detect and analyse relationships between different (groups of) substances or species (variables) and environmental conditions,
- to assist in the interpretation of laboratory and field experiments,
- to identify new research topics and
- to guide water managers to select the optimal strategy for managing a specific water body.

A variety of eutrophication models has been developed and applied for water quality management in the past decades. Models range from purely conceptual to largely determined by data and in practice they are combinations of concepts and data. Eutrophication models may describe the behaviour of one variable in one specific system or they may be very complex. They may form a part of large decision support systems, for example to predict the development of the water quality on a national scale for the next decades. Such models analyse alternative scenarios for the economic and social development or simulate different scenarios for the effect of global climatological changes. Eutrophication models thus differ in spatial and temporal scales and describe different variables. Furthermore, changing objectives, developments in the state of the art of scientific knowledge and improved facilities offered by computer hard and software, cause a continuous adaptation of existing models and the development of completely new models.

Using models carries certain risks. Topics or problems are translated into model objectives; the objectives are reduced to model variables and mathematical equations in computer code. Only a small part and a limited period of the real world can be simplified into a model. Climatological conditions, emissions and interactions with the real world outside the modelled system are reduced to technical boundary conditions and manageable model input. Simplifications and assumptions are necessary and even essential in the modelling process, but they introduce an uncertainty in the precision of the model results. The complexity of eutrophication models describing the various processes as illustrated in Figure 1, increases towards the top of the causal chain, whereas knowledge

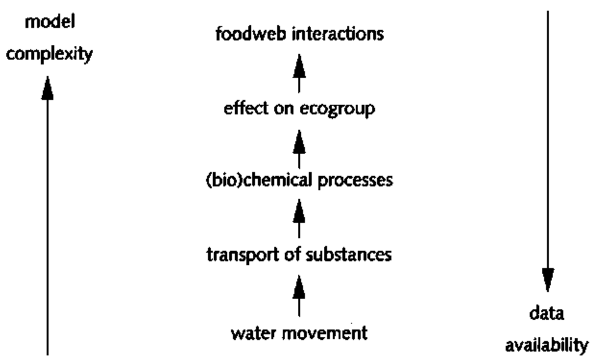


Figure 1 *Model objectives related to complexity and data availability (after Van Beek, 1993).*

and data availability decrease in that direction. A consequence may be that the precision of the results produced by different eutrophication models will also differ and that more sophisticated models, although producing more details, do not necessarily produce better results. A 'very complex' model is not necessarily better than a simple rule of thumb. The precision of the weather forecast with complex models is only slightly better compared to 'the weather of tomorrow will be the same as today'.

This thesis is a reflection and analysis of the credibility of eutrophication models and of their relevance to water quality management. The objective is to extract guidance from previous work for new initiatives on eutrophication model development and application. A number of papers on topics related to the development and application of eutrophication models serve as a background (Appendices). In the extended introduction to these papers, model credibility and acceptability will be stressed and illustrated.

Chapter 2 presents a general approach to eutrophication model development. This approach is based on systems analysis and aims at achieving predictive credibility for the model. The theory is illustrated with examples from the field of eutrophication of surface waters. Chapter 3 focuses on the uncertainties related to models. Furthermore, a distinction is made between credibility and acceptability of a model or its results. Finally, criteria are formulated to evaluate and guide the development and application of eutrophication models. In chapter 4 some of the eutrophication models developed and applied in The Netherlands are evaluated with the help of the criteria from the previous chapter. Furthermore, acceptability of model results is emphasised using interviews with three lake managers. Finally, chapter 5 summarises the conclusions.

2 System analytical approach of eutrophication

Systems analysis is an analytical methodology that helps to identify and select a preferred course of action among several feasible alternatives. It is a logical and systematic approach wherein objectives, assumptions and criteria are clearly defined and specified (Patten, 1971-1976; Biswas, 1976; Young, 1983; Straskraba & Gnauck, 1985; Beck, 1997). Systems analysis is thus used to set up the framework for a meaningful procedure for developing eutrophication management strategies, in which models are a key tool. These models are applied to provide a better understanding of the processes in the water system and to predict the consequences of several alternative courses of action. The framework is visualised in Figure 2 and will be used as a leading thread through this chapter.

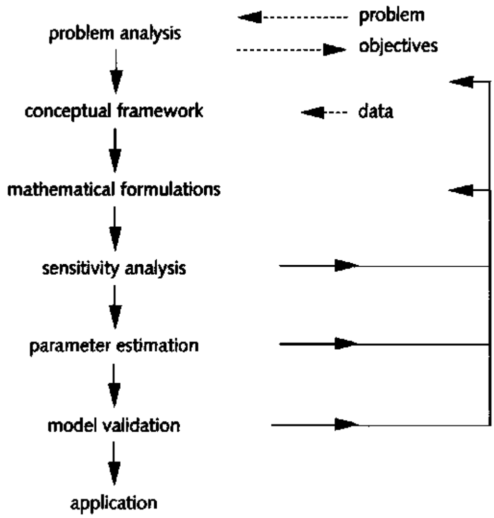


Figure 2 *Stages in systems analysis.*

During problem analysis a general problem is specified and translated into objectives. These objectives are further specified in a conceptual framework that may be formalised into a set of mathematical equations governed by parameters. Next, during the sensitivity analysis, parameter estimation and validation, the model is developed and adjusted to the specific circumstances of the system under consideration using the available data. The system is identified if the patterns of behaviour predicted by the model do in fact resemble observed patterns of behaviour, i.e. if the set of relevant mathematical relationships and the parameters used in these relationships are able to describe the observed behaviour of the system. Finally, the model is applied.

The suggestion may arise that the framework represents a 'one-way-procedure' or a linear process from problem analysis to model application. However, in practice there are many cyclic elements in this framework. Each step may induce adjustments in the previous steps and parts of the framework will have to be repeated before the model may be applied successfully. For example, system identification is represented in Figure 2 as the cyclic procedure to obtain a validated model.

2.1 Problem analysis - formulation of objectives

Eutrophication is still one of the major problems in water quality management in The Netherlands. Problems may be experienced by both the manager (standards are not met, or goals such as the restoration to a former or even pristine situation are difficult to achieve) and the public (the water smells, or the water looks like pea soup). For freshwater lakes, recovery and restoration are often the ultimate targets for managers. Until recently these targets were almost directly translated into a reduction of the phosphorus concentration in lakes by decreasing point source and non-point source emissions or by in-lake measures to reduce the nutrient availability (e.g. Golterman, 1970; Sas, 1989; Jeppesen *et al.*, 1991; Cooke *et al.*, 1993). About a decade ago this 'bottom-up' approach was supplemented by a 'top-down' approach, to accelerate lake recovery by food web manipulations (e.g. Benndorf, 1987; Gulati *et al.*, 1990; Meijer *et al.*, 1994). Nowadays, several other factors than nutrients are recognised to enhance eutrophication. These factors relate to the hydrology, chemistry and ecology of the system.

Perceptions of a problem, global targets of water managers and an enumeration of explaining factors are not enough to set the modeller to work. Problem analysis aims at producing an explicit statement of the objectives. Therefore, the problem should be translated into operational and quantitative objectives and criteria have to be specified in order to identify promising solutions (for example with respect to time and money). As this is seldom common practice, two examples ('Phosphorus' and 'Blue-green algae') may illustrate the procedure for deriving objectives and criteria from a general problem.

Phosphorus example

Suppose the high phosphorus concentration in a specific lake is considered to be a problem, because a certain water quality standard is not met or because it is assumed that ecological recovery necessitates a lower phosphorus concentration. The objective may then be: *The phosphorus concentration in the lake should be less than 50% of the present concentration within five years.* When the target is to reach the water quality standard, this is only an appropriate objective if the present concentration is twice the standard. Otherwise, it should be explicitly stated that the objective is only a first step towards the standard. If 'ecological recovery' is the real target behind this objective then the objective implies that knowledge exists that this target will be approached through this objective. Alternatively, it should be explicitly stated that the objective is only a first step to the ultimate target. In this example the response variable is specified, the reduction level is quantified and measurable and the period allowed to reach this goal is clear. However, no criteria are specified as to the selection of alternative roads leading to possible solutions. In fact design of measures is a separate task. In the first instance a reduction of the external phosphorus loading may be considered, but there are also other options. For example, measures reducing the internal loading of phosphorus from the sediment may be envisaged. A set of management measures and associated costs may be specified, for example: *The phosphorus concentration in the lake should be less than 50% of the present concentration within five years through reducing the external phosphorus loading from point sources or through a more cost-effective in-lake measure.*

A further restriction may be that only some in-lake measures will be studied that have already been carried out successfully in comparable systems. An extension of cost-effectiveness may be that not all alternatives that do not meet the objective will be rejected; a cheap alternative (for example 'do nothing') resulting in the desired phosphorus concentration within ten years may be preferred to an expensive alternative that is effective within three years. Also, criteria for the durability of the result may be specified, as well as criteria for the meteorological conditions for which the objective has to be met, etc.

Design of measures

The choice of measures may be within or outside the modelling context. In the two examples, the design of alternatives seems to be a separate task. Two approaches in the design of measures may be distinguished, one result-orientated ('how do we reach the goal?') and one *feasibility*-orientated ('which measures are realistic?'). In practice, the choice of measures will be determined by previous experiences, feasibility, but also imagination. The use of models may assist this process, but the choice of a specific model or model variables and relations may also restrict the creativity to develop alternatives. For example, it is not surprising that a top-down approach to combat eutrophication effects was not considered in projects that focused on nutrients only (e.g. Hosper, 1978; Hieltjes & Lijklema, 1981; PER, 1982, 1986).

Blue-green algae example

Blooms of blue-green algae may cause nuisance. The public may experience the nuisance because blue-green algae maintain a high turbidity or because they are able to produce toxins under certain conditions. The task of a lake manager may then be to avoid or to get rid of blooms of blue-green algae. This may result in the objective: *Blooms of blue-green algae should not occur in the lake.* It is assumed that (only) blue-green algae cause nuisance when blooming. A further specification or even modelling of the nuisance as such is not considered here. A further specification of 'bloom' is necessary, for example '> 75% of the algal biovolume consists of blue-green algae in samples representative for a period of two weeks'. A comprehensive discussion on the definition of 'harmful blooms' was presented by Smayda (1997).

A first step in the problem analysis may be to identify the conditions that favour blue-green algae. In the next step these conditions have to be compared with the actual conditions in the lake. Finally, management alternatives may be developed in order to meet the (minimal) conditions required for preventing blooms of blue-green algae. The management alternatives may be restricted to nutrient concentrations (e.g. external loading reduction) and mixing regime (e.g. compartmentalisation of the system, artificial mixing, flushing). A more specific objective is: *Determine the conditions in the lake preventing that more than 75% of algal biovolume consists of blue-green algae over a period of two weeks or more and relate these conditions to the nutrient loading to the lake and management alternatives affecting the mixing of the water of the lake.*

Additionally, budget criteria may be specified to limit the feasible alternatives to the most cost-effective ones. Finally, for both this and for the former example, it may be considered to add 'quality conditions' to the model outcomes related to the objective. For example, Rykiel (1996) proposed that the most important variables of dynamic model output fall within the 95% confidence interval of the observations 75% of the time. Several aspects related to (prediction) uncertainty are emphasised in paragraph 3.1.

For the examples given above, the problems are translated into operational and quantitative objectives. However, it is difficult to include and specify a priori all targets and criteria. Neither is it always advisable to impose very rigid specifications as they may limit the possibility of finding an optimal alternative. Moreover, the formulation of objectives does not automatically guarantee that the problem can be solved as the formulated objectives may not cover the entire problem.

2.2 Conceptual framework

In the next stage a conceptual framework is set up to provide information on ways to achieve the objectives defined in the problem analysis. At the end of this stage it will be apparent
a) which variables will be included in the model,
b) which relationships between variables will be included,
c) what spatial schematisation will represent the modelled system,
d) how temporal aspects will be addressed,
e) what kind of data are needed and what interactions with the environment will be considered.
The two examples from the previous paragraph will be used again to illustrate the methodology for deriving a conceptual framework from the problem analysis. The examples will give an overview of aspects that may or may not be included in the framework. Letters refer to the items listed above.

Phosphorus example - The phosphorus concentration in the lake should be less than 50% of the present concentration within five years through reducing the external phosphorus loading from point sources or through a more cost-effective in-lake measure.
a) and b) The response of the lake is delayed by buffer mechanisms, for example elevated internal loading of phosphorus from the sediment as long as no equilibrium with the new external loading is reached (Lijklema, 1983, 1986; Jeppesen *et al.*, 1991; Van der Molen & Boers, 1994; Van der Molen *et al.*, 1998b). Initially, the set of mathematical relationships may be restricted to a simple, dynamic budget model for the phosphorus concentration in the water phase. Internal loading based on empirical data may be included to account for the buffer mechanisms. For some type of measures this approach may be insufficient and, if there are enough data available, the statistical approach may be expanded with a dynamic phosphorus budget model of the sediments (e.g. Lijklema, 1982, 1983; Van der Molen, 1991; Van der Molen *et al.*, 1998b) or more sophisticated sediment phosphorus models, taking into account adsorption characteristics, etc. (e.g. De Rooij, 1991; Smits & Van der Molen, 1993). Consequently, also the phosphorus content of the sediment has to be a variable in the model and maybe also other sediment characteristics. On the other hand, for studying the objective it need not be necessary to include detailed descriptions of, for example, nitrogen dynamics and phytoplankton. Reduced levels of phytoplankton may affect sedimentation of organic matter and consequently the release of phosphorus from the sediments. However, it may be assumed that the primary production of phytoplankton is not affected in the phosphorus concentration range considered (i.e. phosphorus is not limiting). Furthermore, it may be assumed that variations in the nitrogen concentration as a consequence of measures do not affect internal loading of phosphorus nor primary production by phytoplankton.
c) Shallow lakes have often been simplified as completely mixed compartments; horizontal and

vertical gradients are disregarded. Occasionally, series of well mixed compartments are considered. Transport between the compartments may be measured input for the model or calculated by a hydrological model. Horizontal gradients may be considered if, for example, wind driven resuspension determines the phosphorus concentration. A vertical dimension is needed to describe phosphorus behaviour in the sediments.
d) The temporal resolution of the model depends on how the internal loading is described. In fact, a detailed description of internal loading is not the main issue, it only serves to represent the delay in the lake response. If empirical data on internal loading are available and sufficient, the temporal resolution may be up to a year. A more detailed description of the internal loading necessitates a temporal resolution that meets with its dynamics. The total simulation period will be a couple of years, but initialisation of the sediment phosphorus content may require a longer time period.
e) The external phosphorus loading, as well as the hydrological loading, are important input data. The temporal resolution determines the detail needed in the boundary conditions. Because of the enormous amount of phosphorus in the top layer of the sediment compared with the phosphorus in the overlying water, initial conditions for sediment phosphorus will be important for the model behaviour. Data on the cost of the measures are required to meet with the last part of the objective. However, the cost of measures is generally not included in this type of models; they are calculated externally.

Blue-green algae example - Determine the conditions in the lake preventing that more than 75% of algal biovolume consists of blue-green algae over a period of two weeks or more and relate these conditions to the nutrient loading to the lake and management alternatives related to mixing of the water of the lake.
a) Phytoplankton will be the key variable. In order to describe competition, several algal species need to be described. The objective expresses algal biomass in biovolume, because this unit is related to the routine measurements, but existing models generally use carbon units as output variable. Furthermore, a choice has to be made whether to describe nutrients within algal cells with a fixed ratio to carbon or to use a variable cell stochiometry (Droop, 1973; Zevenboom *et al.*, 1982). Initially, availability of nutrients, light and grazing pressure may be modelled as input conditions instead of variables, because this reduces model complexity significantly (e.g. Huisman, 1997). This approach assumes that availability of nutrients and light and grazing pressure are not coupled to the model variables, although this is not the case. Therefore, this simplifying assumption should be verified during the modelling process. If the assumption is valid with respect to the modelling objectives, the relationship between for example the available nutrients and grazing pressure on the one hand and management alternatives on the other may be studied by using other models.
b) Initially, it may be assumed that only processes related to phytoplankton growth and succession will be modelled and that nutrients, light and top-down control are inputs. The model thus aims at determining the conditions that favour blue-green algae or defining their habitat in relation to the availability of nutrients and light. Furthermore, competition with other species and top-down control have to be studied. Vertical transport of phytoplankton in relation to light availability is known to be a potential key factor in the blue-green algal competition strategy (Reynolds, 1987; Ibelings, 1992). Horizontal transport, driven by wind, is important for the formation of floating layers of decaying blue-green algae which cause a lot of nuisance. Because blooms and not floating layers are of primary interest, horizontal transport may probably be neglected. 'Memory effects'

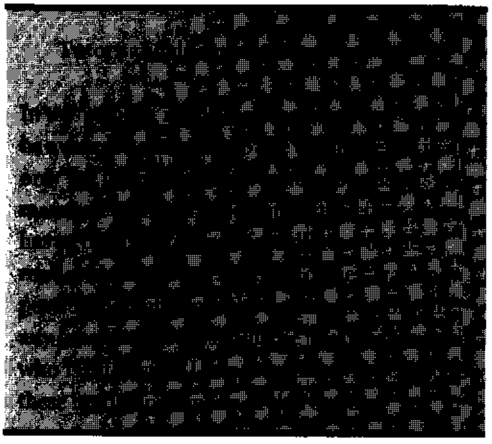
may be important. Survival of blue-green algae on the sediments or persistence of blue-green algae during the winter may be important processes as they affect blue-green algal dominance in the following summer (Reeders *et al.*, 1998).

c) Vertical transport requires a detailed description of the vertical gradients. In general, the model complexity increases considerably with the introduction of an extra spatial dimension. Therefore, 'pseudo' gradients are sometimes introduced. For example, light availability decreases exponentially with depth in the water column, but in models light availability is often averaged by integration over the water column to prevent the introduction of the vertical dimension. If nutrient concentrations are assumed as inputs, sediments do not have to be included in the model schematisation.

d) Variations of meteorological conditions within a day may be important. For example, changing light conditions may affect luxury uptake of phosphorus in the algal cells. The total calculation time can be a year or less. However, when memory effects are included, for example survival of blue-green algae on the sediments during the winter, the simulation period should be longer to account for the initial conditions.

e) As mentioned before, nutrient availability, irradiance and grazing pressure may be input variables. Probably these data are required with a high temporal resolution.

The examples illustrate that the choice of variables, spatial dimensions and temporal aspects can to some extent be derived from the objectives. For both examples it may be possible to develop a quantitative model. At this moment a 'go-no-go' moment has to be considered, taking into account the amount of time and money available to develop the model (or to adjust an existing model) and to apply the model. Moreover, if the objective is not precise enough, or if the data availability is a problem, one might return to the previous stage in systems analysis.



2.3 Data

Data availability may affect the chosen model complexity (see Figure 1 in chapter 1). Figure 3 illustrates several types of data and their relationship to the steps in systems analysis. The data are grouped as follows:

– Spatial dimensions. Data have to be supplied for the model schematisation. For example, the depth (distribution) of the system in case the water level is not a variable in the model. If the water level is an (input) variable, a relationship between volume and water level may be required for calculating the surface area of the system. Increasingly GIS applications are being combined with models to supply spatial dimensions.

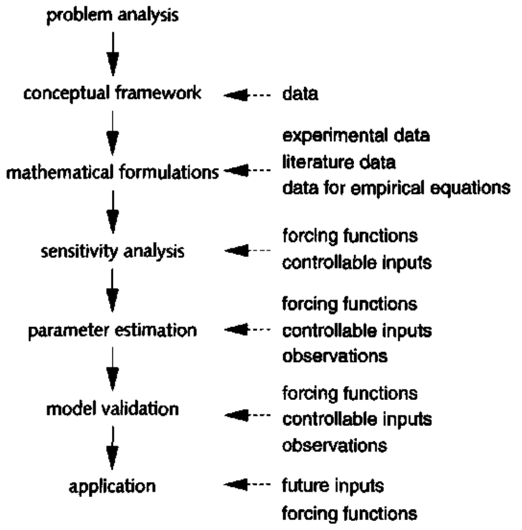


Figure 3 *Types of data and their relationship to systems analysis.*

– Input data. The model input data can be categorised in controllable and uncontrollable data. The first set is also called instrument variables, manageable data or steering factors, as they can be influenced by management. External phosphorus loading from point sources is an example. The second set of input data is also called non-manageable data or environmental conditions. Irradiation and wind speed are examples of uncontrollable data. 'Boundary conditions' and 'forcing functions' are terms used for both controllable and uncontrollable inputs. Furthermore, there is a difference between input data from the present/past and future inputs. Future inputs are needed for scenarios. Controllable future input data are determined by expected developments or optional management strategies. Uncontrollable future input data may be used directly from past records or generated from noise functions and random generators. In some cases uncontrollable future input consists of extreme conditions instead of average conditions, for example to represent maximal risk situations.

– Information to set up relationships between the model variables. The choice of the variables and the type of relationship between variables can be derived from a statistical analysis based on a large data set. Also, a priori knowledge (accepted theories, laws) may be used to formulate model equations. However, the choice for a specific type of relation (for example zero or first order kinetics) may also be based on data. Furthermore, different relationships may be used for specific concentration ranges or a threshold temperature is introduced for a specific process. In these cases additional data are necessary to determine the ranges or the threshold level.

– Parameter values for the model equations are data that may be obtained from literature, from especially designed experiments and independent measurements and/or by parameter estimation techniques using the model on the basis of a set of observations (paragraph 2.6).

– Initial values of the model state variables. Variables in differential equations need to be initiated. The initial values may be derived from measurements or may be approximated by simulation as the steady state that the model reaches with constant input data.
– Observations, not previously used for parameter estimation, to check if the behaviour of the model (output) corresponds with the observations from the real system.

In the ideal situation all information is readily available. In many cases, however, the model structure and (some of the) parameters will not be based on appropriate data, but on assumptions. Sometimes additional information may be gathered during the model building procedure and the model framework may be used for the identification of additional experiments or measurements. However, limited data availability may be such a constraint that the objectives have to be adjusted and another conceptual framework has to be derived from the objectives (see previous paragraphs). On the other hand, data availability is only one of the factors determining model complexity. As will be discussed in the next chapter, a model more complex than justified from the point of view of data availability, may be defensible.

2.4 Mathematical formulations

In the conceptual framework (paragraph 2.2) an outline of the model is generated; the objectives are made explicit and are quantitative as much as possible. During model construction the modelled system is defined as a set of state variables and the functional relationships between these variables are formulated as equations, including the parameters. The set of mathematical formulations are often referred to as the 'model'. A more general definition of a model is 'a representation of the essential aspects of an existing system (or a system to be constructed) which presents knowledge of that system in a usable form' (Eykhoff, 1974). However, in this definition it is suggested that for each system another model is needed. The 'model' fits only a specific application. In this thesis a model refers to a tool that can be used for several systems, which of course share some important characteristics. Furthermore, in this thesis only quantitative, mathematical models, related to eutrophication and as applied in freshwater quality management, will be studied. Table 1 summarises several ways in which these quantitative, mathematical models may be classified.

Eutrophication models, as well as other environmental models, have certain characteristics in common, but do not necessarily fit into one of the categories of Table 1. On the other hand, eutrophication models and their mathematical formulations may be characterised by the evaluation of these categories with the items summarised in paragraph 2.2:
a) In eutrophication models, state variables are generally aggregates for many individuals or different species and their behaviour is therefore only partly predictable. The choice of state variables is not unequivocal. For example, zooplankton may be described as a state variable or as a boundary condition.
b) Certain parts of the structure of eutrophication models are more or less generally accepted, but other parts are heavily debated.
c) The spatial dimensions of the system can not be defined unambiguously. For example, an average depth does not account for the real depth distribution of a lake.

Table 1. *Classifications of model types (Clarke, 1973; Beck, 1981; Van Straten, 1986; Jørgensen, 1994; Scholten & Van der Tol, 1994).*

Hard models	Characterised by sufficient and generally accepted theories and hypotheses. The dimensions of the system are clear and the input data are measurable and controllable.
Soft models	Characterised by highly unpredictable variables. The dimensions of the system in which the variables operate are vague and the hypotheses on the behaviour of the variables are not well established. In general, there are complex, non-linear relationships between input and output.
Conceptual models	Based on causal relationships between inputs, state variables and outputs, using the principles of mass and/or energy conservation. Theory-orientated.
Statistical models	Based on the reduction of large numbers of observations, without necessarily applying general laws and principles. Data-orientated. In a strict sense, cluster analyses relate (groups of) variables to each other and no a priori knowledge or general laws or principles are introduced (black box models). In practice, some assumptions or even expected causal relations are used to find relationships in a large data set with observations. Also referred to as empirical models.
Deterministic models	Predicted values are computed free from random variations.
Stochastic models	Predicted values take into account probability distributions of the input data, parameters and the equations.
Dynamic models	State variables are a function of time.
Static models	State variables are not time-depending.

d) Several topics in the field of eutrophication require insight in system dynamics. For example, one might be interested in the response time of a system after a measure has been applied. On the other hand, there are several examples of static, often statistical, models.

e) Most environmental models are both theory and data orientated and may be regarded as 'grey box models'. The question if a water quality model is conceptual or statistical depends on whether emphasis is put on the theory or the data.

From theory to data

Theory and data seem to have had a different impact on the development of sediment models, aiming at describing the internal phosphorus loading in lakes. In the eighties theory predominated. The complete equilibrium chemistry and several sediment layers were included in models (e.g. De Rooij, 1991). Later on, both the schematisation and the chemistry were simplified and adjusted to data availability (Smits & Van der Molen, 1993), but due to lack of data these models could only be applied to a limited number of lakes. Van der Molen & Boers (1994) started with the available data on internal loading and sediment characteristics (ratio total phosphorus to iron) and developed a simple statistical model.

In conclusion, the type of eutrophication model depends largely on what is required from the objective. After a conceptual framework is derived from the problem analysis and the framework is translated into mathematical formulations, the system is further identified by sensitivity analysis, parameter estimation and model validation. These three steps will be distinguished and discussed in the following paragraphs.

2.5 Sensitivity analysis

Model structure identification, the procedure of finding the set of relevant mathematical relationships, may be seen as the more or less successful completion of the repetitive cycle from the conceptual framework to the sensitivity analysis (see also Figure 2). Sensitivity analysis provides a measure to determine the effect of variations in input data, parameter values and initial values of state variables on the state variables. Sensitivity may be expressed with (1).

$$G' = \mathrm{d}x \,/\, \mathrm{d}q \qquad (1)$$

with G' = measure of sensitivity
x = state variable
q = input data, parameter or initial condition

To overcome the problem of different dimension of G', (1) may be altered into (2).

$$G = (\mathrm{d}x/x) \,/\, (\mathrm{d}q/q) \qquad (2)$$

with G = measure of sensitivity (-)

The sensitivity of state variables for the input gives an indication of the accuracy that is required for the input data. The sensitivity of state variables for the model parameters helps to select the parameters that should be involved in the parameter estimation procedure. Only parameters for which state variables are sensitive can be adjusted by parameter estimation. If, on the other hand, state variables are not sensitive at all to parameters or if state variables show extreme perturbations on changing parameter values, identifiability of (parts of) the model structure may be questioned for the assumed excitation. In that case the modelling concept may be reconsidered, as illustrated in Figure 2. However, the sensitivity of the state variables for a parameter, and therefore the (lack of) identifiability, can only be determined for a given input range and for the given initial conditions. A similar reasoning holds for the sensitivity of state variables for the input or the initial conditions.

Input data, parameter values and initial values for the state variables should be varied within realistic ranges. Frequently, the excitation is a fixed percentage of the nominal value. For measured input data, parameters and initial conditions, the excitation may be based upon (a percentage of) their standard deviation. Furthermore, a minimum and a maximum variation of the default value of inputs, parameters and initial conditions should be analysed, as the response of the state variables is generally not linear. Moreover, in dynamic models, x, q and therefore the sensitivity may vary with time. For example, the sensitivity for the initial conditions will be highest in the beginning of the simulation and the sensitivity of the model variables for zooplankton grazing parameters will be noticeable mostly during periods when zooplankton is present. Accordingly, the sensitivity may be determined at a specific moment or averaged for (a part of) the simulation period. For larger models the sensitivity may be determined for each state variable separately or an 'average' sensitivity may be determined. The sensitivity may further be determined for a single point in the parameter space or the sensitivity may be studied for combined variations in parameter values, inputs and initial conditions. With respect to this, also parameter correlation should be addressed. For example, an increase in both the growth rate and settling rate of phytoplankton may result in a similar phytoplankton biomass.

In this paragraph state variables are implicitly assumed to be also the output variables and the sensitivity of the state variables is assumed to equal the sensitivity of output variables. In fact, only sensitivity of the output variables is important with respect to the model objectives. For example, dissolved and particulate phosphorus may be state variables, but total phosphorus the output variable. If variation in one state variable is compensated by the other, then the output variable will be less sensitive to the analysed excitation.

2.6 Parameter estimation

Figure 4 is a pictorial representation of the relationships between the real system behaviour, the observations and the model (Mankin *et al.*, 1977; Scholten & Van der Tol, 1994). The objective of models is to approximate certain characteristics of the real world as closely as possible, or, in terms of Figure 4, to find a best accordance between model output (*Y*) and the system behaviour (*S*). However, the behaviour of the system is only approximated by the observations (*O*) and the procedures used to find the best accordance between model output and system behaviour fit, in fact, model output to the observations. Given a specific model structure, improvement of the agreement between model output and a set of observations is performed by parameter estimation.

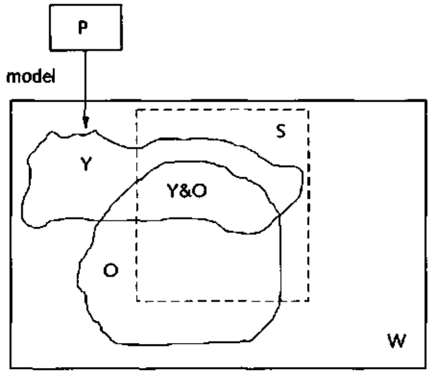


Figure 4 *Two-dimensional projection of the model parameters (P), the domain of all possible behaviour patterns (W), the system behaviour (S), model output (Y) and observations (O).*

Sensitivity analysis identifies the parameters that have the largest impact on the state variables. If parameter uncertainty is not a priori taken into account in the sensitivity analysis, parameters for which the model is sensitive may be distinguished in parameters with narrow and wide confidence limits. The latter are the most important parameters to focus on during parameter estimation. Parameters that hardly influence the state variables cannot be estimated from observations. In that case well-known parameters can be fixed, while the use of non-sensitive parameters with wide confidence limits should be reconsidered (Thomann, 1982; Van Straten, 1986).

Figure 4 illustrates that part of the observations do not reflect the real system behaviour. Errors in sampling or in the analytical procedure of a chemical compound may serve as examples. Fitting model output and observations may take such errors into account. The Figure also illustrates that the model output will be partly out of the range of the real system behaviour, but also partly inside, without overlap with observations. In this way, parts of the model output that cannot be checked by observations may still be valuable.

Calibration

Many authors define calibration as the procedure to find the best accordance between computed and observed output variables by variation of some selected parameters. In this sense calibration equals parameter estimation. According to Beck (1981) the word calibration is misleading as 'it suggests an instrument whose design is complete and whose structure is beyond further argument'. He, and with him several others, uses calibration in a way that resembles system identification. As both interpretations can be described satisfactory by parameter estimation and system identification, the word calibration will be ignored as far as possible. If 'calibrated model' is used, the latter meaning of the word calibration is referred to.

Maximisation of the intersection between model output and observations, as illustrated in Figure 4 by Y&O, is equivalent to the minimisation of the difference between model output and observations (3).

Minimise $D(p)$ $$D(p) = D\{Y_t(p), O_t\}_{t=1,\ldots,T} \quad (3)$$

with D = discrepancy measure
Y = model output
O = observation
p = parameter (from parameter set P)
t = time (from 1 to T)

One single model output variable may be compared with observations or several model output variables may be compared simultaneously. In the latter case Y and O in (3) become vectors and weight factors may be introduced in the optimisation procedure. For example, the total phosphorus concentration may be more important compared with total nitrogen and chlorophyll-a, when measures are considered to reduce the phosphorus loading. Besides the choice of the model output variables and the assignment of weight factors the way in which model output is considered is also important. All the individual observations may be considered or only extreme values, averages or values above a certain standard, etc. These and several more choices are arbitrary aspects of the parameter estimation procedure. Van der Molen & Pintér (1993; Appendix A) discussed the pros and cons of several possibilities of selecting a discrepancy measure.

Several parameter estimation procedures are available. Most procedures can handle a limited number of parameters only and need specified ranges in which the parameters may vary. Some procedures yield information for correlation between parameters. Therefore, apart from improving the fit between model output and observations, parameter estimation may be a tool to test the model structure. Even after minimisation of $D(p)$, there will be a discrepancy between model output and observations (4).

$$Y_t - O_t = e_t \qquad (4)$$

with e_t = residue

Analysis of the behaviour of this residue, for example for periodicity or a relationship with input conditions or state variables, may also be used to reconsider the model formulations (Figure 2). Beck (1983) and Young (1983) advocate the use of a recursive parameter estimation procedure to test the model structure. This is a parameter estimation for each individual observation, that may yield variable parameter values. Significant changes in these parameter values may trace omissions in the model structure.

2.7 Validation

In literature the term validation is used in different ways (see Rykiel (1996) for an overview). According to many authors validation is a test for accepting or rejecting the model. Consequently, in the first stages of system identification performance criteria have to be specified to be able to make an objective judgement on the validity of the model. However, in practice some parts (variables) of the model may behave satisfactorily, whereas other simulated variables deviate seriously from observations. Or, for a particular water body the model is precise but transfer to another system or generalisation fails. Also, the model results may resemble observations pretty well for a certain period (season), while for another period it does not. Of course, all these aspects can be included in criteria, but completeness is very difficult and the weight of the different factors may be perceived differently according to the specific application. In theory, a model can only be invalidated, because proving that its results are conform reality everywhere and always is impossible. A 'valid model' does probably not exist in this conventional, rigorous sense (Oreskes *et al.*, 1994; Beck, 1997).

Hence, a distinction is made between validation and acceptance or rejection of the model (results). Validation is defined as a test for determining the degree of agreement between a model and the observations of the real system (e.g. Goodall, 1972; Mankin *et al.*, 1977; Jørgensen, 1994). For this, the parts of the model that are (in)valid should be designated, the input range that is used should be specified, the water body and the period for which the validation test is performed should be described, etc. Instead of a general acceptance or rejection the model is thus tested for its suitability, taking into consideration the intended use of the model (see also chapter 3).

For both parameter estimation and model validation, model output is compared with observations. Therefore, equation 3 may also be used for model validation and the remarks made in paragraph 2.6 considering the application of this equation also hold for model validation. Nevertheless, for the validation of a model it is necessary to use a set of observations independent from the data used for the estimation of model parameters. These may be observations on the same system for similar conditions (also referred to as data-splitting), or on the same system but under different conditions and system behaviour, or on a different compartment of the same system and finally on another system. Successful validation of these subsequent stages increases the robustness of the model. Validation based on data-splitting was found to be unsatisfactory if the period related to the data set is shorter than the hydraulic retention time of the water body (Simons & Lam, 1980). As a rule of thumb, with respect to parameter estimation and validation the data set should be larger than the largest time constant of the system and the frequency of observations should be less than the smallest time constant of the system.

> **Verification**
>
> **Verification and validation are often used as synonyms (e.g. Thomann, 1982; Thomann & Mueller, 1987). However, for example Straškraba & Gnauck (1985) distinguished two modes of model testing. Verification is the comparison of simulation results and observations, while validation is a test for similarity between model and system behaviour. Sometimes verification is also used for model structure identification. The use of the term verification will be avoided in this thesis.**

Prior to the validation of a model with independent data or in the case of independent data being insufficient or even absent, a more qualitative validation procedure may be considered (Thomann, 1982; Rykiel, 1996). Experts may be asked if the model behaviour is reasonable (face validity) or if they can discriminate between observed system behaviour and model output. Furthermore, model output may be compared with output from other models. Validation may also focus on specific events, without further quantification (event validity). For example, does the model describe a switch from diatoms to green algae at the proper time?

The type of test used for validation depends on the availability of data and the understanding of the system (Rykiel, 1996). Lack of data and poor understanding of the processes imply that only qualitative tests (for example face validity and event validity) can be used for validating the modelling concept. If, on the other hand, a large number of data and knowledge of the processes is available, a quantitative validation is possible. Two other combinations can be envisaged: many data and little understanding or vice versa. In the first situation a statistical validation may be performed, while in the latter only a qualitative validation is possible.

Analogously to parameter estimation, validation of a model may be performed given its structure and parameters. However, as for parameter estimation, model validation may also be used for testing the model structure, i.e. the choice of the state variables and the relationships between the variables. For example, validation under a combination of extreme input conditions may reveal information on the model structure. If the model does not simulate the set of observations satisfactorily, the process of systems analysis has to be restarted at least from the point where the mathematical model was derived from the conceptual framework. As mentioned previously, identification of the model structure and optimisation of the parameter values may be regarded as cyclic processes.

2.8 Application

If the results of the validation are acceptable, the model can be applied. This final stage of systems analysis consists of the generation and evaluation of alternatives or scenarios. This will be illustrated using the examples introduced in paragraph 2.1.

Phosphorus example - The phosphorus concentration in the lake should be less than 50% of the present concentration within five years through reducing the external phosphorus loading from point sources or through a more cost-effective in-lake measure.
In simulations the external phosphorus loading may be varied to see if this alone is sufficient to meet the objective. If so, several realistic alternatives for the reduction of the external loading may be analysed with regard to cost-effectiveness. Special attention should be paid to the difference between a decrease in the phosphorus concentration in the input water and a reduction of both the loading with water and phosphorus, for example in case the diversion of a brooklet is an option. In the latter case the hydraulic retention time will increase and this may result in a relatively larger contribution of processes, such as internal loading, to the concentration in the lake. Besides reduction of the external loading, the effect of other measures should be studied. For example, flushing with water low in phosphorus or chemical fixation of sediment phosphorus (e.g. Hosper, 1984; Jagtman *et al.*, 1992; Cooke *et al.*, 1993). A combination of reduced external loading with such additional measures may be an cost-effective alternative. If there is no realistic scenario that meets the objective, the model might give the explanation for this and the marginal benefits of slackening the objective may be assessed.

The phosphorus concentration also depends on meteorological conditions (Portielje & Van der Molen, 1998). As the conditions for which the alternatives must be analysed are not specified, the effect of promising alternatives may be studied for different meteorological conditions. Results of the sensitivity analysis, the sensitivity of output variables for realistic excitations of input, parameters and initial values of state variables, may be used to specify some measure of 'accuracy' for the different alternatives.

Blue-green algae example - Determine the conditions in the lake preventing that more than 75% of algal biovolume consists of blue-green algae over a period of two weeks or more and relate these conditions with the nutrient loading to the lake management alternatives related to mixing of the lake.
Suppose a 'habitat' model is developed that describes the phytoplankton composition as a function of input conditions, such as nutrient concentration, light availability, mixing and grazing. These boundary conditions may then be varied individually in order to analyse the appearance of blue-green algal blooms. Next, combinations of boundary conditions may be studied. If the model predicts the occurrence of blooms of blue-green algae the possibilities to change controllable input data in such a way that undesired blooms are prevented, have to be examined. Finally, the feasibility of the measures has to be studied considering technical aspects, budget constraints, accuracy of the model predictions, acceptability, etc. The artificial mixing of lake Nieuwe Meer may serve as an example of a possible solution preventing nuisance blooms of *Microcystis* (Visser *et al.*, 1996).

3 Credibility and acceptability of eutrophication models

In the previous chapter systems analysis is presented as a methodology to tackle complex problems with the help of simplifying models. Between the lines it has been noticed that uncertainties are present in almost every stage of this process. Uncertainties are due to errors in the different types of data (paragraph 2.3). In this chapter the uncertainties are categorised. Correct assessment of uncertainties is related to the credibility of a model or to the credibility of a specific application of a model. Credibility is explicitly distinguished from acceptability. A set of criteria will be given for evaluating or guiding model development and application.

3.1 Uncertainty

The errors in the data are propagated in the model structure, the model parameters and in the model predictions (e.g. Beck, 1983, 1987; Morgan, 1990; Rowe, 1994). The relationship between system identification, prediction and uncertainties is illustrated in Figure 5.

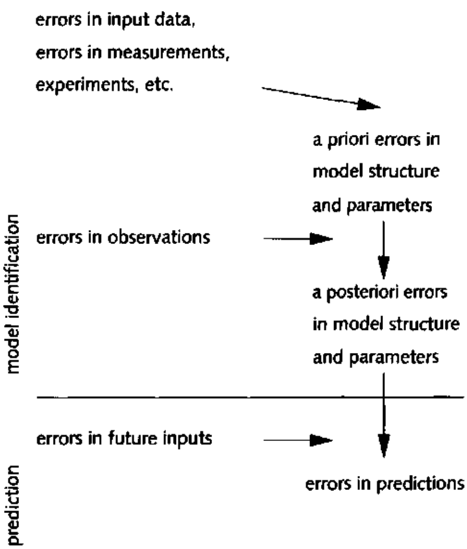


Figure 5 *Errors in system identification and prediction.*

A priori errors in the model structure and parameters are introduced during the first stages of systems analysis. Errors in observations are added during parameter estimation and during the validation of the model, resulting in a posteriori errors in the model structure and parameters. These a posteriori errors are not necessarily larger than a priori errors as the different sources of errors may partly neutralise each other. Finally, the a posteriori errors are combined with errors in future inputs, leading to errors in the predictions during the final stage of systems analysis. In this paragraph uncertainty in the model structure, the model parameters and in the model predictions are highlighted.

uncertainty in the model structure

Uncertainty in the model structure is categorised in

- errors in the mathematical formulations,
- errors due to spatial and temporal aggregations,
- errors due to omission and/or aggregation of variables, processes and parameters,
- numerical errors.

Mathematical formulations may be derived from theory or from data (paragraph 2.4). There is often more than one 'accepted theory'. For example the light efficiency curves for phytoplankton, expressing the relationship between light irradiance and gross production rates, may be described with or without a reduced production rate at very high irradiance levels (photo-inhibition). Without sufficient specific measurements an inferior formulation might be chosen in the model. Another example is whether a constant or variable nutrient content in phytoplankton cells should be used. If a fixed ratio is used the model structure may be appropriate for certain conditions but for other situations it may not be possible to find a suitable parametrisation that will reproduce the observed behaviour of the system. Mathematical formulations derived from data only (statistical models) also contain errors. For example, a Monod-type and a logistic-type of growth function for phytoplankton are difficult to discriminate when data on phytoplankton biomass and nutrients are analysed. As not the process itself is measured, but only its results, the exact form of the equation can only be estimated.

> **Error and uncertainty**
>
> **The true value of an entity (a variable, parameter or the model structure) is always unknown. An estimate may be obtained by measurements or by model calculations. The error is then defined as the deviation between the true value and the estimate. Similarly, a model structure and a prediction contain errors. Uncertainty is the statistical description of the errors; it is a measure of the closeness to the true value.**

In the conceptual framework several types of aggregation have to be made. The spatial and temporal dimensions of the model are necessarily aggregations and are therefore inherently affected by errors. For example, a lake may be simplified as one well mixed compartment and vertical gradients in the water column may be disregarded. Within the well mixed water body the model variables are homogeneously distributed by definition. If, for example, surface blooms of toxic blue-green algae are modelled as homogeneously distributed, model parameters may have to be adjusted to unrealistic values in order to reproduce the phytoplankton composition. With respect to aggregation in time, time steps range from hours, days or even years. The choice of the integration time-step may be a source of errors related to the temporal dimensions as variations within these aggregated time intervals are disregarded.

Individual species and individuals within a population of different ages or stages of development are mostly aggregated within their functional groups. For example, phytoplankton can be modelled as one group, as a restricted number of classes or as many different species;

zooplankton can be divided into random and selective feeders or in many groups or species each with its characteristic grazing behaviour. Analogously, total phosphorus represents several fractions. Furthermore, physical, chemical and biological processes and parameters are lumped in the model. For example, net sedimentation of particles is often formulated instead of gross sedimentation and resuspension and the mineralisation of organic matter represented by a first order decay rate is a simplification of many processes in the bacterial community. Aggregation, i.e. lumping of several variables, processes, etc. into a single determinant, is the consequence of omissions, i.e. describing only those determinants that are expected to be dominating. In many models epiphytic plankton or macrophytes are neglected and only phytoplankton is assumed to contribute to the primary production. Again, all these aggregations and omissions may be necessary simplifications, but they introduce errors in the model structure (O'Neill & Rust, 1979).

Finally, numerical errors introduce uncertainties. For example, the value of a state variable has to be adjusted to a minimum level if a solution of a differential equation becomes equal to zero or below zero.

uncertainty in the model parameters

Parameter values are obtained from different sources, for example especially designed controlled experiments or independent measurements, literature data and/or parameter estimation. In all cases errors are introduced. By repeating experiments or measurements the error can be quantified. Occasionally ranges are given for parameters obtained from literature data, so their uncertainties may be quantified. Sometimes parameters derived from experiments and literature are adjusted by parameter estimation, but for other parameters estimation may even be the only way to generate a suitable parameter value.

During parameter estimation and during the validation of the model errors in the controllable and the uncontrollable input and in the observations are propagated in the model structure and/or the estimated model parameters. As illustrated in Figure 4 (paragraph 2.6), part of the observations do not reflect the real system. Besides errors in sampling and analytical procedures, errors are introduced if observations represent a large spatial compartment and a long time interval instead of the measured momentary, point value. Errors due to spatial inhomogeneity may be reduced by increasing the number of sampling locations and, consequently, the number of observations or by analysing a (weighted) mixture of water from different locations. Errors due to temporal variations in the set of observations may be reduced by more frequent sampling and/or by measuring in periods with relatively large variations in the observed variable.

Furthermore, errors in the assumed or measured initial conditions of the system will affect parameter estimation. This is especially a problem if initial conditions determine the state variables for a long period relative to the total simulation time. For example, in eutrophic, shallow lakes the pool of phosphorus in the top layer of the sediments is much larger than the amount of phosphorus in the overlying water. Therefore, estimation of model parameters by comparison of the observed and the modelled phosphorus concentrations in the water may result in unrealistic parameter values if the initial sediment phosphorus content deviates significantly from the real sediment composition. In the parameter estimation procedure the model parameters will also be adjusted to account for the wrong sediment composition.

Another source of errors in the estimated parameter values is the parameter estimation procedure itself. In general, parameter estimation of environmental models will not reveal one single, optimal set of parameters. The procedure may end in local minima for the discrepancy measure instead of the optimal solution, especially if several parameters are involved simultaneously in the parameter estimation procedure (Pintér, 1990). With respect to the parameter estimation procedure, the unconscious bias of the modeller by the selection of the specific procedure and the discrepancy measure, the acceptation/rejection of possible erroneous observations and the appraisal of the model results should be mentioned here (Salt, 1983; Van der Molen & Pintér, 1993).

uncertainty in the model predictions

During system identification errors in the several types of data are propagated in the structure of the model and in the parameters of the model. This a posteriori error (Figure 5) will further be propagated into the model predictions. Frequently, the a posteriori error may be accepted, provided that it is taken into consideration when predictions are made. Alternatively, the model may be 'adjusted' to neutralise (part of) the a posteriori error, if the error is quantified and related to an input or a state variable. For example an empirically derived parameter related to the temperature may be introduced if the a posteriori error is correlated with temperature.

Two other types of errors will be introduced when the model is used for predictions. First, future input variables obviously contain uncertainties. The uncertainties in model predictions depend on errors in the assumptions concerning the controllable and uncontrollable inputs for future scenarios. Second, prediction errors may be caused by significant changes in the structure of the system compared with the system for which the model was validated. In the next paragraph the issue of system changes will be highlighted.

It could be argued whether it matters that predictions are highly uncertain, as long as the decisions informed by these predictions are insensitive to such uncertainty (Beven, 1993; Beck, 1997). In other words, if the same course of action would still be ranked first among various alternatives, despite all uncertainties, the decision could be said to be robust. Accordingly, the evaluation of uncertainties shifts from predictions to decisions. Aspects related to this will be mentioned in chapter 4.

3.2 Extrapolative and projective prediction

Theoretically, a validated model is valid for the ranges of input and observations that have been used during system identification. However, even within those ranges a correct prediction is not guaranteed. Structural changes of the system may appear within the defined input range. For example, an existing species, not included in the model structure, may become important to the food web or unexpected species may be introduced due to some random event. These system changes are more likely to occur, when the model is applied outside the ranges used for model identification. In fact, the goal of eutrophication models is often to find the conditions that may cause a switch of the system to a different state (Van Straten, 1998). In other words, the model is forced to 'look' outside its ranges. Using the terms from Figure 4 (paragraph 2.6), the situation can be envisaged by Figure 6.

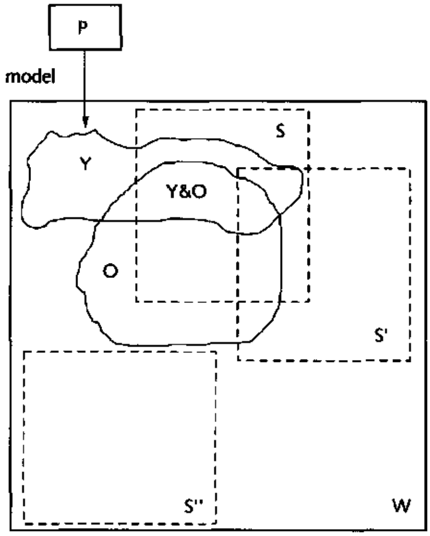


Figure 6 *Two-dimensional projection of a structural change in system behaviour from S to S' and S''.*

In the worst case both previous observations and the model results have no relationship to the new system. In other cases however, the model output is still capable of reproducing, at least partly, the new system. The model output might even be the best guess for predicting a different state of the system, because data are not yet available. Van der Molen & Boers (1996; Appendix D) evaluated a study of Boers *et al.* (1991), in which they predicted the food web structure in Lake Wolderwijd, The Netherlands, after food web manipulation. Some predictions of Boers *et al.* (1991) were justified by observations, but other estimated pools and fluxes deviated significantly from the observations. Moreover, benthic algae appeared to be an important group in the food web after the food web manipulation, but they were not included in the study of Boers *et al.* (1991) at all.

Structural system changes

An example of a structural change in a system is the appearance of the unexpected species *Neomysis integer* after a system perturbation imposed by management. This freshwater shrimp was found in large numbers after a drastic reduction of the fish biomass to stimulate zooplankton grazing. Possibly, the shrimp predated on zooplankton, therefore decreasing the effect of the management measure (Meijer *et al.*, 1994). Another example of a structural system change is the reappearance of macrophytes as primary producers in eutrophic lakes (Meijer *et al.*, 1994; Van der Molen & Boers, 1996; Noordhuis, 1997). These system changes can be observed and may be incorporated by adjustments in the model structure or model input. However, some system changes may even not be detected. For example, the illegal removal of a top predator by poaching may result in an unexplainable change in the food web structure.

Following Van Straten & Keesman (1991) and Van Straten (1998), the following procedures may be used for improving model predictions in such situations:

- speculate about possible changes in the system and express these changes in terms of modifications of existing parameters, based on expert knowledge,
- speculate about potentially important processes in the future and incorporate them into the model structure, even though the value of parameters may not yet be known very well,
- incorporate concepts of self-learning and adaptation in the model; for instance Straskraba (1979), Los (1980), Los & Brinkman (1988) and Straskraba (1994) summarise ecological principles that may be used in model development and Jørgensen (1988, 1992) promotes the concept of exergy to incorporate flexibility and selection in ecological models.

If, for example, zooplankton grazing has not been observed in the past then inclusion of zooplankton in the model is based on experiences in other systems. No suitable information is available yet for the parametrisation of zooplankton grazing, except from information from other systems. The model contains highly uncertain surplus content, is probably over-parametrisized and will probably not be identified. For this situation Beck (1981, 1983) postulated the paradigm, that either a well calibrated model can be used to predict the state of the system under significantly changed conditions, with the risk of predicting the wrong future with great precision, or non-calibrated processes can be added to the model, with the chance that the future is predicted correctly, but with a significant uncertainty. Beck (1983) suggests splitting the model structure in an identifiable part and a more speculative part. The former part has to be identified or falsified, while the latter part has to be addressed in a more cautious way.

Van Straten (1986) introduced the terms 'extrapolative prediction' for the forecasts based on calibrated, but inadequate models and 'projective prediction' for the forecasts with models that cannot be calibrated, but may be more adequate. Exploring the possibilities as mentioned above, is a form of 'educated speculation' with models (Van Straten & Keesman, 1991) and speculation becomes an identified step in the prediction process. However, this step is a maverick in uncertainty analysis.

3.3 Credibility and acceptability

In chapter 1 'the credibility of eutrophication models and their relevance to water quality management' is postulated as a central issue. Young (1983) uses model credibilty in a rigid way as 'a property which depends upon success in all phases of the model-building procedure'. Rykiel (1996) defines credibility as 'a sufficient degree of belief in the validity of the model to justify its use for research and decision making' and relates credibility to the amount of knowledge available, the purpose of the model and the consequences of any decision based on it. He considers credibility as a subjective qualitative judgement. Here the definition of credibility will be restricted; *a eutrophication model becomes 'credible', if the model went through the procedure of systems analysis* (chapter 2) *and if the uncertainties involved in predictions have been considered* (paragraph 3.1). Analogous to the definition of Rykiel, credibility is characterised as sufficient insight in the precision of the different parts of the model and in the range in which the model can be applied quantitatively or qualitatively (paragraph 3.2). On the other hand, the definition used here is less subjective and qualitative compared with the approach used by Rykiel (1996) and may be seen as a certification ('good modelling practice') rather than a qualification. In the extreme situation the model may still be credible, while parts of the model suffer from large uncertainties. Nevertheless, the suitability of the model is addressed in 'credibility', as validation is an identified step in assigning credibility. For example, a model may be in general credible and, specifically, be valid for e.g. total phosphorus, but not for chlorophyll-a. Or the model may be only valid for the summer period. Thomann (1982) proposed a 'post-audit' validation in determining credibility, that is an evaluation of the model results after implementation of these results.

A crucial point in this thesis is the distinction between credibility and acceptability. After credibility of the model has been specified, the model may be either accepted or rejected. *So, credibility is the more technical appropriateness of the model and acceptability is the perception of the manager of its practical value.* This distinction between credibility and acceptability is made because it represents common practice. Systems analysis is performed mostly by modellers, but model results are used by managers. Managers have to judge if model results and the uncertainties are acceptable to be used in decision making.

Managers and modellers

The underlying thought of the positioning of credibility and acceptability on the one hand and the actors on the other, is illustrated in Figure 7. Credibility should be addressed by the modeller, while acceptability is the responsibility of the manager. Acceptability may also play a role between the operational managers and policy makers and between the governors of an institution and other parties concerned. However, no emphasis is put on these interactions here. Although the situation illustrated in Figure 7 is common practice, it represents a rather simplified situation. There may be several other interactions, for example caused by an identified role of specialists besides modellers and managers (Thomann, 1982), with an interactive planning process as the extreme situation (e.g. Van Rooy, 1997).

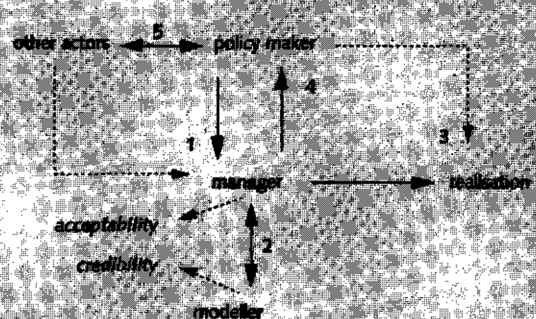


Figure 7. Positioning of credibility and acceptability and the actors in the modelling study. The (operational) managers derive objectives from 'the field' or objectives may be imposed by policy makers (1). Next, managers initiate a modelling study (2). Finally, results are put into effect (3) or passed on to policy makers (4), who may negotiate with other actors (5) before implementation of the modelling outcomes.

Model credibility and, specifically, validity for the relevant output variables may contribute to the acceptability of a model or its results. However, other considerations involved with acceptability may have a strategic, managerial or psychosocial background. For example, managers may have to set off rejection of model results against making a decision in another way or postponing the decision. Managers may decide that the model results are the best guess, irrespective of uncertainties or lack of significant discrimination between alternative courses of action. The Dutch surface water standard for the total phosphorus concentration was derived from a statistical model for stagnant, shallow lakes and based on summer averaged data (CUWVO, 1987). The use of this standard for annual means as well and also for other types of freshwater systems was a strategic decision (Ministerie van Verkeer en Waterstaat, 1989). The scientific basis of more than one standard was set off against the confusion introduced with more standards for the total phosphorus concentration. Recent scientific evidence for the inappropriateness of the eutrophication standards for certain types of freshwater lakes (Van der Molen *et al.*, 1998c) did not lead to adopting the suggested refinements as well. Furthermore, model results may be accepted because other actors approve of the results. For research institutes of three ministries, representing agriculture, water management and environmental aspects, this lack of acceptability has contributed to further co-operation and a restart of systems analysis, as illustrated in Figure 2, aiming at a joint development of a model describing nutrient losses from soils to surface water and groundwater (Boers *et al.*, 1996).

Some of the strategic aspects are related to managerial aspects. Examples of managerial aspects affecting acceptability relate to constraints in time and money. Credibility may be set off against the time and money that has been put in the modelling exercise or the time and money that still is available for a certain deadline. The time schedule of national policy cycles is generally fixed. The choice for existing models or the development of (partly) new models used for national policy calculations, is restricted by this time schedule. Furthermore, if the resources are not sufficient to carry out an uncertainty analysis, results may be accepted 'because the model has proven to be successful in another application'. Sometimes absolute values of the prediction are recognised to be 'wrong', but relative differences between alternatives are accepted as model results. A psychosocial aspect may be that managers tend to be 'result-orientated' and may accept model results leading to action easier compared with model results that do not ask for action. On the other hand, lack of public support may more likely lead to acceptation of model results that imply no action. Decisions of managers may be biased by (un)popular alternatives. For example, measurements and model results indicate that buffer strips between agricultural land and surface water have little potential in reducing nutrient emissions from agriculture in The Netherlands at the present high input rates (Orleans *et al.*, 1994; Boers, 1996; Van der Molen *et al.*, 1998a). Because buffer strips are attractive for natural values of an area, they are still frequently mentioned as a promising measure to reduce nutrient emissions from agriculture (e.g. Boers *et al.*, 1995).

3.4 Criteria for eutrophication models

In the previous paragraph the relationship and the difference between credibility and acceptability are stressed. The ideal situation is a credible model, that is accepted based on its performance during the systems analysis. However, this is not always true. The examples, mentioned in the previous paragraph, of more or less irrational acceptance are widespread in practice. In this paragraph criteria are set up to determine model credibility and to specify the acceptability of the model. The distinction is based on the presumption that as validation results and uncertainties should be specified by modellers, analogously acceptance criteria should be made explicit by managers. The criteria, listed below, are extracted from chapter 2 and paragraph 3.1 for formalising credibility and from paragraph 3.3 for formalising acceptability. The criteria will be applied in the following chapter.

Criteria for credibility.

1a *Objectives* of the model are specified and the choice of state variables is in agreement with these objectives.

1b *Dimensions* of the modelled system and *aggregations* in time and space meet the objectives and the availability of data.

1c The available *data* are utilised sufficiently and the system identification is not hampered by lack of input data and observations; uncertainties in the data are considered.

1d During sensitivity analysis, parameter estimation and validation, the appropriateness of the *model structure* is examined.

1e Model *parameters* are fixed at well documented values or are properly estimated.

1f Model *validation* is based on an independent set of observations and the results are quantified and related to the objectives.

1g The *uncertainties* in model structure, model parameters and model predictions are addressed and quantified to a certain extent.

Criteria for acceptability.

2a The *motivation* for the initiation of a modelling project is known.

2b *Constraints* in time and money for model development and application are specified.

2c *Arguments* for approval of the model (results) are made explicit.

2d *Consequences* of the use of the model (results) are discussed.

As already stated, the criteria for credibility and acceptability may be applied to the development and application of a new model as well as to the model results when applying an existing model. The criteria 2a and 2b will generally precede criteria 1a - 1g, while criteria 2c and 2d are relevant after the criteria for credibility. Some of the criteria can be quantified, but others, especially the criteria for acceptability, are rather qualitative. Finally, the criteria for acceptability focus on acceptance of (the results of) a credible or non-credible model, but rejection should be treated in a similar way.

4 Application of criteria for credibility and acceptability

In the previous chapter credibility and acceptability were explicitly distinguished. Furthermore, criteria for credibility and acceptability were derived from the theory of systems analysis, from a discussion on uncertainties and from some illustrations of motivations related to the (dis)approval of model results. In this chapter these criteria will be used to evaluate three models. The examination of these models may be regarded as an a posteriori analysis of a modeller in a context of management and policy making. Because of the perspective of the reviewer, credibility will be emphasised. Therefore, next, acceptability will be highlighted using interviews with lake managers about projects in which eutrophication models were involved.

4.1 A posteriori review of three eutrophication models

Because causes and appearances of and restoration from eutrophication are complex issues, managers have often been assisted by models in making decisions. Three different types of eutrophication models, applied to freshwater lakes in The Netherlands, will be evaluated in the following paragraphs. These models and their applications will be reviewed with respect to the criteria for credibility and acceptability, as mentioned in paragraph 3.4. The model described in Van der Molen & Boers (1994; Appendix B) will be studied as an example of a statistical model

Eutrophication

Water management in The Netherlands has focused on water quantity for centuries. Protection against water, land reclamation, but also transport of water to dry areas were major issues in the previous centuries. Frequent occurrences of epidemic diseases like cholera and typhoid did not result in a broad consideration of water quality aspects until the 19th century. At the end of the 19th century enhanced population pressure and industrial activities resulted in serious, large scale water quality problems. In the beginning of the 20th century water quality deterioration by organic pollution was recognised. After the second World War, the effect of toxic substances in the water became evident. It lasted until the latter half of the sixties before eutrophication was considered a large scale problem in The Netherlands and elsewhere (Gorham, 1964; Vollenweider, 1968; Golterman, 1970).

Eutrophication is the process of nutrient enrichment. It is a natural ageing process, accompanied by changes in species and community structure and ultimately resulting in a transformation into a terrestrial ecosystem. This slow process of ageing can be greatly accelerated by human interventions. Human activities are often responsible for an over-fertilisation of rivers, lakes and coastal areas. This cultural eutrophication results in an abundant growth of macrophytes or in aquatic ecosystems dominated by phytoplankton. In the first stage of eutrophication, primary production will be stimulated. This is beneficial to the biomass of some species and may be profitable for commercial fisheries. With progressive eutrophication, however, the species diversity will decrease and both ecological values, recreational and economic use will be negatively affected.

(paragraph 4.1.1). The model described in Van der Molen *et al.* (1998b; Appendix E) will be used as an example of a simple, deterministic model (paragraph 4.1.2), whereas the eutrophication model described in Van der Molen *et al.* (1994; Appendix C) will be studied as an example of a more complex, deterministic model (paragraph 4.1.3).

4.1.1 Statistical relationships describing the phosphorus concentration in lakes

Surveys of water quality data, mainly nutrients, chlorophyll-a and transparency, resulted in the first statistical models in The Netherlands (CUWVO, 1976; Hosper, 1978; CUWVO, 1980). These broad analyses of water quality have been repeated and extended afterwards. In CUWVO (1980) only phosphorus was analysed, whereas CUWVO (1987) also included nitrogen. In the latter algal species were distinguished in blue-green algae and others. Portielje & Van der Molen (1998) also distinguished *Microcystis* from filamentous blue-green algae and examined the effect of zooplankton grazing, macrophytes coverage and fish densities on the nutrient - chlorophyll-a relationships.

Apart from the relationships between water quality variables and biological variables the relationships between the hydraulic and external nutrient loading and the concentration in the water were studied in these projects as well. Similar statistical studies were carried out elsewhere (e.g. Vollenweider, 1969, 1975; Reckhow, 1979). The relationships tested were based on several assumptions, for example a well mixed lake and equilibrium between water and sediments. To increase the applicability of these statistical models for situations with a changed external phosphorus loading, Van der Molen & Boers (1994) included internal phosphorus loading and sediment characteristics into the model structure. The result of this study will be used here as an example of the application of criteria for credibility and acceptability on a statistical model.

Statistical model of Van der Molen & Boers

The statistical model of Van der Molen & Boers (1994; described in detail in Appendix B) aims at predicting the summer mean total phosphorus concentration in shallow freshwater lakes from the hydraulic loading and the external and internal nutrient loading. Two types of relationships were tested: one was derived from the mass balance of total phosphorus for the system and the other was determined by linear multiple regression. The external phosphorus loading determined the phosphorus concentration in lakes without a reduced loading ('mass balance based model', r^2=0.88, n=23, p<0.01), whereas the internal phosphorus loading determined the phosphorus concentration in lakes after a reduction of the external loading ('mass balance based model', r^2=0.88, n=10, p<0.01). Moreover, the internal loading was replaced by sediment characteristics to allow prediction of the phosphorus concentration after reduction of the external loading, using measured data prior to measures in combination with expected loading reductions. The correlation coefficients became only slightly lower, but the predictive power of the model decreased ('multiple regression model', r^2=0.68 and 0.82 for sediment phosphorus and the phosphorus to iron ratio respectively, n=8, p>0.05).

1a objectives[1]

The study aimed at statistical relationships capable of predicting the phosphorus concentration in shallow freshwater lakes with and without a reduced external phosphorus loading. The single state variable is the summer mean total phosphorus concentration in the lake (gP m^{-3}), corresponding with the modelling objective. The independent variables of the statistical models are the hydraulic loading (m y^{-1}), the external and internal phosphorus loading (gP m^{-2} y^{-1}), and the total phosphorus content (gP kg^{-1} dry weight) or the total phosphorus to total iron ratio (gP gFe^{-1}) in the sediment.

1b dimensions and aggregations

Lakes included in the analysis were assumed to be well mixed and to be represented by their sampling location(s). If detailed spatial information was available, data of different locations were averaged using surface area weighting factors. Because the summer is the most important period for lake eutrophication effects, the state variable and the input variables were averaged for the period from April to September. No information was supplied for the number of individual data available to calculate summer mean values and for the method of averaging. In case of non-equidistant data, simple direct averaging may introduce serious errors.

'Cause and effect', here loading and concentration, were implicitly assumed to occur without time lags. This is questionable, because the relative contribution of the external and internal nutrient loading to the concentration in the lake will change depending on the number of years after a loading reduction measure (Lijklema, 1983, 1986; Jeppesen *et al.*, 1991) and on several other factors, such as the hydraulic residence time and meteorological conditions. The schematisation of the lakes and the spatial and temporal aggregation of the variables justify only objectives without much detail in time and space.

1c data

Data of 49 lakes were gathered from literature, including three lakes that appeared twice in the data set (observations before and after measures). For only 27 lakes data were available for all variables. Summer means and the frequency distributions of the variables were presented in the paper. Logarithms of the data were used to decrease the effect of extreme values of the variables, although not all variables were log-normally distributed. Moreover, data on the hydraulic loading and the external phosphorus loading of two lakes were omitted because of the a priori expected large uncertainties, while other data of these lakes were used. Loading data of two other systems were not used, because the hydraulic retention time indicated that the system resembled a stream rather than a lake (Boers & Van der Molen, 1992).

No quantitative information on errors in the data was reported. The phosphorus concentration was supposed to be a 'relatively reliable' variable, while the largest uncertainty was expected in the external phosphorus loading and the hydraulic loading. However, information on errors in the concentration and loading data is available. The analytical error in total phosphorus is estimated at 7 - 19% (see paragraph 4.1.3). For intensively studied lakes, PER (1982) and PER (1986) reported that errors in the dominating sources of the external phosphorus loading range from 5 - 20%. For smaller

1 The model (application) is examined with the criteria mentioned in paragraph 3.4. The numbers and subtitles refer to these criteria.

contributors this range is considerably larger. Furthermore, the studies mentioned did not take the analytical errors into account. For lakes without this detailed mass balance information, the errors will be larger. In general, loading errors are indeed expected to be larger than errors in the total phosphorus concentration, including spatial errors and temporal errors. Data on internal loading may contain larger errors compared with the external phosphorus and hydraulic loading, taking into account the lower frequency of sampling compared with the total phosphorus concentration, the larger spatial heterogeneity of sediments compared with the overlying water, differences in analytical methods used and errors related to artefacts when estimating in situ fluxes with laboratory experiments. Errors in sediment characteristics due to spatial heterogeneity will probably dominate errors due to temporal variation and analytical errors. Therefore, depending on the way spatial heterogeneity is accounted for, errors in sediment data may be as large as errors in loading data.

1d model structure

Two model structures were applied. One was adopted from previous studies (Vollenweider, 1975) and based on the mass balance of total phosphorus for the system. The other was generated by multiple regression of combinations of independent variables. Correlation coefficients between calculated and measured total phosphorus were slightly higher for the multiple regression model compared with the model derived from the mass balance.

1e, 1f, 1g parameters, validation and uncertainties

Parameter values were estimated by regression. All data were used for the parameter estimation, so the relationships were not validated on independent data. The number of data was too small to make subsets to be able to estimate parameters and validate the model on independent data sets. Significance levels of the correlation analysis give an indication for the predictive power of the statistical models. However, the use of tests for significance levels for non-(log-)normally distributed variables is questionable if the data-set is smaller than about 15 cases. Furthermore, significance levels may be misleadingly high for relationships with both external and internal loading as independent variables, because these variables are inter-correlated. The statistics of the residuals between calculated and measured concentrations and the significance levels of the parameter values were not discussed.

An analysis by Portielje & Van der Molen (1998) confirmed the relationship between internal loading and the phosphorus concentration in the lake. However, application of the model on independent data resulted in significant deviations between measured and calculated values (Rijsdijk, 1995). Accordingly, the objective to develop a quantitative statistical model for the prediction of the summer mean phosphorus concentration before and after a decrease of the external loading could not yet be met fully.

2a, 2b motivation and constraints

The motivation of the study was to gather data on external and internal phosphorus loading and to make this information available to lake managers dealing with internal phosphorus loading when planning or evaluating a measure to decrease the external phosphorus loading. No constraints with respect to time or money have been specified for the development of the model. In fact, there were none of these constraints other than the personal responsibility of the authors.

2c, 2d arguments and consequences
No arguments were specified for acceptance/rejection of the models. The statistical properties (r^2) of the relationships were implicitly assumed to be indicative for their credibility. Consequences of the use of statistical models that do not take into account the internal loading are discussed for lakes with and without reduction of the external phosphorus loading. Furthermore, Van der Molen & Boers (1994) postponed the issue of acceptance of their relationships by explicitly stating that more data are needed to validate the model and thus enhance its credibility. Consequently, the paper has been cited to illustrate that knowledge about controlling factors of internal loading has increased rapidly in recent years, but that it is still inadequate for precise predictions (e.g. Kleeberg & Kozerski, 1997).

4.1.2 A mass balance model for the phosphorus content of the sediment

Mass balance models in eutrophicaton studies generally refer to simple, deterministic models, aiming at describing the phosphorus concentration in the water (e.g. Hieltjes & Lijklema, 1981; PER, 1986; Van Straten, 1986). In some cases the sediment phosphorus was explicitly taken into account in the modelling concept. Information on changes in the phosphorus pool of sediments as a function of external conditions is indispensable for long term predictions on shallow lake trophic conditions. Therefore, Van der Molen *et al.* (1998b) utilised a mass balance model for the sediment phosphorus pool. This paper will be used as an example to illustrate the application of criteria for credibility and acceptability on a simple deterministic model.

1a objectives
The objective of the model is to make long term predictions on the trophic state of a shallow lake. The main state variable is the total phosphorus content of the top layer of the sediment, representing the trophic state of the lake. The modelling objective is related to long time scales (at least several years) and therefore no relationship was described between sediment phosphorus and, for example, the phosphorus concentration or trophic indicators in the overlying water. Trophic indicators such as the biomass of phytoplankton and macrophytes were considered to be independent of the availability of nutrients beyond a threshold concentration. Furthermore, no other nutrient than phosphorus was taken into account. Literature data reveal that trophic conditions in freshwater lakes are mostly determined by phosphorus (Hutchinson, 1957; Golterman, 1973; Jeppesen *et al.*, 1991) and that available phosphorus is strongly coupled with phosphorus in sediments (Boström *et al.*, 1982; Lijklema, 1983, 1986).

1b dimensions and aggregations
The model was applied to Lake Veluwe, The Netherlands. The lake was schematised into two compartments, because two areas can be distinguished with remarkable differences in depth, soil type and seepage/infiltration fluxes. The dynamic model was only applied for the deeper part of the lake, because equilibrium was assumed between import and export of phosphorus to the sediment top layer in the shallow part of the lake. The time step of the model equations was one year. Variations within a year were considered to be neither important from the point of view of the

Mass balance model for sediment phosphorus

The mass balance model for sediment phosphorus utilised by Van der Molen *et al.* (1998b; described in detail in Appendix E) aims at predicting long term developments in the trophic state of shallow lakes. The total phosphorus (P) content in the top layer of the sediment was assumed to represent the trophic state of the lake. The model is basically:

sediment-P = f(initial state, surface in and outflow of P, seepage/infiltration of P, burial)

The model was applied to Lake Veluwe, The Netherlands, schematised into two different compartments. Net sedimentation was assumed to occur in the muddy, deeper part of the lake only. Sediment phosphorus in this compartment is further affected by infiltration and burial with sandy material resuspended in the other part of the lake. Some results reconstructing the past and predicting the future sediment phosphorus content and fluxes under present boundary conditions are depicted in the figure below.

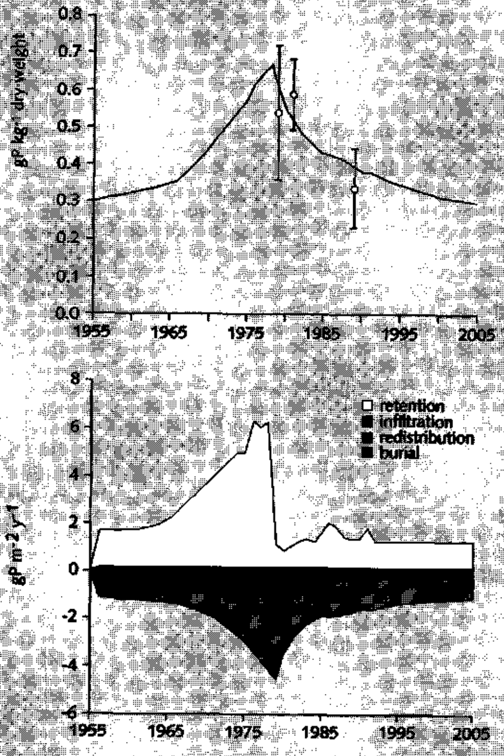


Figure 8 *Calculated (line) and observed (circle and standard deviation) phosphorus pool (top) and calculated fluxes (bottom) in the top layer of the sediment of the muddy part of Lake Veluwe, The Netherlands.*

objective nor significant for this state variable. Because most model input variables were available monthly or yearly and because sediment characteristics were measured at a limited number of locations, this temporal and spatial aggregation is justified by the data availability. Furthermore, these choices seem appropriate to model the sediment phosphorus pool.

1c data

The input variables for the model were the surface inflow and outflow of phosphorus, seepage, infiltration and burial. Surface inflow and outflow of phosphorus were derived from detailed mass balances. Seepage and infiltration flow rates were estimated with another model and were assumed to be constant. The phosphorus concentration in seepage was based on measurements. The phosphorus concentration in the infiltrating water was coupled to the total phosphorus content in the top layer by means of a Langmuir adsorption isotherm. Redistribution of solids in the lake was assumed to dominate net external loading or formation of solids. Thus, erosion in the shallow part of the lake was the source of burial in the deeper part. Most input variables and all model parameters were derived from measurements. This was possible, because extensive monitoring and research programs were carried out in Lake Veluwe (PER, 1986; Brinkman & Van Raaphorst, 1986; Van Ballegooijen & Van der Molen, 1994; Danen-Louwerse *et al.*, 1996; Noordhuis, 1997). Available standard deviations were presented, but uncertainties were not specified for all data.

1d model structure

A sensitivity analysis was carried out by variation of individual inputs and parameters and by combined variation of the two factors for which the sediment phosphorus content was most sensitive. The variation of the individual input data and parameters was fixed (10%) and the combined variation was derived from estimated standard deviations around the nominal values. As the variations resulted in significant, but not extreme variations in the state variable, the model was considered to be able to describe the main processes and to contain neither redundancy nor hypersensitivity in its structure.

1e, 1f parameters and validation

Because all parameters were based on measurements, no parameter estimation was carried out. The simulated phosphorus pool in the sediment prior to the reduction of the external loading in 1979 increased as was expected; this may be seen as an example of a qualitative validation (paragraph 2.7). Further validation was performed by visual comparison of the model output and observations carried out during three years in the eighties. The model output falls within the ranges of measurements for all these years. The result of the validation is not quantified, but considered to be satisfactory with respect to the objective and for the projected medium long term range (2005).

1g uncertainties

Uncertainties in the model predictions were addressed using the results of the sensitivity analysis. The sensitivity of the sediment phosphorus content for variations in inputs and parameters for the period after validation, indicates the possible output range in the predictions. However, the imposed variations in inputs and parameters were not directly derived from quantified uncertainties in these inputs and parameters. Furthermore, only for the most sensitive inputs (infiltration and burial) the effect of combined variations was studied.

The study demonstrated both quantitative and qualitative use of the model. Input ranges for which the model was applied were specified. The concept of the model was considered to be applicable for other shallow lakes, although specific characteristics for Lake Veluwe may not occur elsewhere.

2a, 2b motivation and constraints

The motivation of this project was to fill the gap between the statistical models, not suitable for predictions on the trophic state of lakes, and available deterministic models. The latter were considered to be 'too big' for the restricted objectives defined here. Besides, the available model DBS (see also next paragraph) was considered to be inadequate for this purpose as one nutrient fraction was not properly incorporated to the sediment composition after sedimentation (other organic nutrients in Figure 9, paragraph 4.1.3). Moreover, the model discussed here focused on long time scales and was therefore regarded supplementary to the other models.

No constraints in time and money were specified for the model application described by Van der Molen *et al.* (1998b). However, the model was not developed 'from scratch' but derived from a model presented by Lijklema (1982), because that concept already described the main features with respect to the objectives.

2c, 2d arguments and consequences

Arguments for (dis)approval of the model were not specified in advance, but acceptability of the results may be positively affected by the credibility of the modelling approach based on many data. The model has been used to study two alternatives. Results for the default calculations and for two scenarios in the future were interpreted qualitatively, implying that comparative use of the results is more appropriate than absolute use. However, the model outcomes were considered to be sufficiently accurate to hypothesise about future developments of the lake: ecological recovery will continue under present boundary conditions. This conclusion has contributed to the decision of the lake managers to give low priority to additional measures, such as food web manipulation, to speed up lake recovery (T.H. Helmerhorst, personal communication).

4.1.3 A complex model for nutrients and phytoplankton

Relatively simple deterministic models, as the one discussed in the previous paragraph, have been expanded into two directions. First, several lumped processes have been described in more detail, for example the chemistry of phosphorus and algal growth kinetics (e.g. Los & Brinkman, 1988; Smits & Van der Molen, 1993). Second, several 'new' variables and processes, such as zooplankton grazing on phytoplankton, have been included in models to be better equipped to answer specific management questions and to meet changed conditions in the field. These 'expanded' models, or more popular 'complex' models, are generally still based on the conservation of mass. Well-known examples of complex, deterministic models can be found in Patten (1971-1976), Canale (1976) and Jørgensen (1983).

In The Netherlands, the development of large, deterministic eutrophication models started in the latter half of the seventies with the initiation of the WABASIM project (WAter BASIn Models). The general objective of this project was to develop ecological and water quality models to support decision making in water policy. Initially the models focused on the Delta area. In the eighties attention was paid to other systems as well. For eutrophication the model JSBACH (Joint Simulation of Biology And Chemistry) was developed. In 1990 this model was replaced by the eutrophication model DBS (DELWAQ-BLOOM-SWITCH). This model contained less chemistry but described more functional groups (e.g. bottom algae) compared with its predecessor. The application of DBS to Lake Veluwe, as described by Van der Molen *et al.* (1994; see also Appendix C) will be used as the third example, illustrating the application of criteria for credibility and acceptability to a complex deterministic model.

1a objectives

The paper describes an application of DBS to illustrate the possibilities of the model, but it does not address specific managerial problems. The general objective of DBS is to increase understanding of eutrophication processes and to be an operational tool for decision making by water managers. These two goals are explicitly coupled: decision making will improve when the use of the model

DBS

The goal of DBS is to increase understanding of the eutrophication process and to be an operational tool for decision making. Rather than 'complex', the model may be described as 'large'; the model contains about 45 state variables and 17 files with parameters in the water, the sediment and a boundary layer (Figure 9). Time-variable inputs are the hydraulic in and outflows, nutrient loading specified for several fractions, irradiation, water temperature, background extinction and grazing rates. Calculations are carried out with a time step depending on the rate of the fastest processes. Output of all variables and fluxes can be produced daily. The model may be applied to one compartment, or to a network of compartments. Initially, DBS was applied to freshwater lakes, but later on (parts of) the model was also used for rivers, estuaries and oceans.

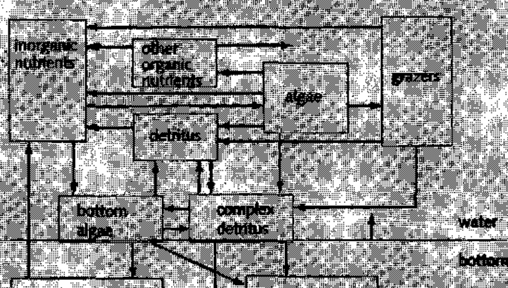


Figure 9 Main model variables and fluxes in DBS (derived from Van der Molen et al., 1994).

enhances knowledge of the system. The variety of topics that may be derived from this general objective necessitates a large number of variables and processes in the model.

1b, 1c dimensions, aggregations and data

The lake was considered to be one homogeneously mixed compartment. Detailed chloride measurements on 23 locations in the lake confirmed that the overlying water is well mixed (PER, 1986). In periods without flushing the standard deviation in the chloride concentration was 2.2% and 4.6% (respectively 31-10-1979 and 22-09-1980). Shortly after the start of flushing the standard deviation increased to 17% (14-11-1979). The flushing water contains relatively high chloride concentrations compared with other hydraulic loads (precipitation, streams, seepage). Two weeks later the standard deviation decreased again to 12% (30-11-1979) and remained more or less constant during the following months with continuously flushing (13%, 29-02-1980). A gradient could be observed with higher concentrations near the pumping station. One month after stopping the flushing the standard deviation had already decreased to 3.7% (27-05-1980). The overlying water is well mixed, but the sediments of the two parts of the lake differ significantly (see previous paragraph). Consequently, in later applications of DBS to Lake Veluwe the lake was divided in two or three compartments. This, however, introduced the need for a highly uncertain exchange between the compartments. The spatial dimensions can be justified by the availability of relevant data, but the use of one 'averaged' compartment is questionable in the perspective of the objective of improving knowledge of the system.

Water quality measurements from a routine sampling station in the deeper part of the lake were available either weekly or biweekly. Ortho-phosphate flux measurements on sediment cores were available for a few locations. Measurements for the most important inputs and outputs in the water balance and for the major external nutrient loads were available on a weekly basis. For smaller contributors weekly input was created by linear interpolation of monthly or even six-monthly data. The simulation period was ten years (1978 - 1987), but one year (1984) was skipped because of lack of input data. Uncertainties in the input data, the initial conditions and the observations were not addressed. However, because of the extensive monitoring programme, mass balance data are accurate compared with most other lake studies (see also paragraph 4.1.1) and many variables were measured with a relatively high frequency.

1d model structure

The sensitivity for the model parameters and the inputs, the admissible ranges of these parameters and testing results of several parts of the model have been reported elsewhere (e.g. Los *et al.*, 1988; Smits & Van der Molen, 1993; Van Ballegooijen & Van der Molen, 1993; Delft Hydraulics, 1994; Los *et al.*, 1994). During these exercises the appropriateness of the model structure was considered. Modifications were introduced, for example extension of the sediment - water module (Smits & Van der Molen, 1993) and addition of grazing (Mooij, 1990; Van der Vat, 1995) and primary production by benthic algae. For these modules testing was not possible or restricted by lack of data, so the model structure could not be identified completely. However, these modules were incorporated because these processes were a priori expected to become important in describing (future) functioning of the lake ecosystem. On the other hand, possible simplifications to the model structure for this particular application were not considered.

1e parameters

In previous studies parameters were estimated visually and without an explicit calibration objective (see 1d for references). A procedure described in Los *et al.* (1994) proposes starting with tuning chloride, followed by chlorophyll-a and finally dissolved and total nutrient concentrations. Three types of parameters are distinguished: fixed, system independent parameters, and fixed, system dependent parameters, and variable, system dependent parameters (Delft Hydraulics, 1994). The first two groups are relatively well known from literature, from experimental data and from previous modelling studies. The fixed, system dependent parameters vary only with the choice of specific model variables such as algal species. The third group consists of about 25 parameters which are especially important for parameter estimation. In a network application for Rijnland only six parameters were adjusted (Delft Hydraulics, 1994). Nevertheless, the identification of the model parameters of DBS is hardly possible; parameter correlation causes different combinations of parameter values (e.g. sedimentation of phosphorus and diffusion controlled release of phosphorus from sediments) to result in comparable outputs.

1f validation

In Van der Molen *et al.* (1994) the model application is presented as a validation, although also some parameters in the sediment - water module were adjusted in this study. The validation was carried out by visual comparison of model results and observations for several water quality variables and for one flux, the release of ortho-phosphate from the sediment. 'Agreement between computed and observed concentrations', reproduction of ' the trend over this ten year period' and timing of specific system changes were mentioned as validation criteria. There is overlap in the ranges of model output and observations for the nutrients and chlorophyll-a. However, uncertainties in both the model results and the observations were not addressed. To make up for this deficiency, some quantitative properties of the validation, not presented in the paper, as well as some analytical errors, are presented in Table 2.

The errors in the most important model variables are larger than the measurement errors. However, apart from these measurement errors also errors in sampling, storage, preparation and representative spatial distribution have to be taken into account when comparing model results with observations. Furthermore, timing errors are important. These errors are partly explained by loss of temporal resolution in the model input and the use of weekly averaged model output and observations. For example, visual comparison indicates that model results and observations of nitrate agree satisfactory (Figure 7 in Appendix C), while the relative errors are large. In conclusion, model errors for the main output variables of some 10% - 40% may be acceptable considering the inherent errors in the observations. Errors in cyanobacterial carbon are significantly larger. Observations of cyanobacteria are based on algal counts and multiplication factors for biomass to carbon ratios. It is more realistic to use these observations in a qualitative way (see paragraph 2.7), for example the presence of cyanobacteria (yes/no) or the timing of the increase and decrease of the population.

The validation of the model is only valid for the input ranges used in the study. DBS was developed for turbid, eutrophic lakes and the model is not adequate for lakes with a relatively large impact of macrophytes or higher trophic levels. This may be illustrated with an a posteriori analysis of another application of the model to Lake Veluwe (Van der Molen *et al.*, 1993). According to the

Table 2 *Number of observations (n), correlation coefficient (r^2) between weekly averaged observations (O_t) and model results (Y_t), root mean square error ($RMS = \sqrt{\{\Sigma(Y_t-O_t)^2 / N\}}$) and average absolute error ($AAE = \Sigma\ |Y_t-O_t| / N$), both absolute and relative to the average observed value (AOV) of the specific variable for the application of DBS to Lake Veluwe 1978 - 1983 as described in Appendix C. Standard deviation of observations within one laboratory (S_r) and between several laboratories (S_R) averaged over 1996 - 1998; Boekholt (1996a, b, c, d, 1997a, b, 1998a, b, c).*

Variable	n (-)	r^2 (-)	RMS ($g\ m^{-3}$) or (dm^{-1})	RMS/ AOV (%)	AAE (mg/l) or (dm^{-1})	AAE / AOV (%)	S_r (%)	S_R (%)
Chloride	219	0.88	16.9	9.95	13.7	8.05	1.76	4.65
Total phosphorus	240	0.78	0.076	34.5	0.060	27.3	6.85	18.6
Nitrate nitrogen	225	0.72	0.50	65.1	0.31	40.7	2.14	5.56
Kjeldahl nitrogen	232	0.62	0.73	28.5	0.59	22.8	13.8	31.9
Chlorophyll-a	213	0.59	59.1	38.7	46.3	30.3	6.99	14.6
Cyanobacterial carbon	134	0.46	4.05	72.9	3.26	58.8		
Transparency (reciprocal)	198	0.51	1.35	35.4	1.07	27.9		

model a summer mean total phosphorus concentration below 0.08 g m^{-3}, a water quality standard of the lake manager, was not achievable with 'best practical means'. However, in recent years lower concentrations were frequently observed without significant changes in the external load (Portielje & Van der Molen, 1998). This may be due to an underestimated change in trophic state as argued by Van der Molen *et al.* (1998b) and/or due to the return of submerged macrophytes. Portielje & Van der Molen (1998) confirmed that macrophytes cause both bottom-up (competition for nutrients) and top-down effects (e.g. allellopathy and protection for grazers) on the chlorophyll-a concentration.

1g uncertainties

It is argued here that both the model structure and the model parameters cannot be identified unambiguously. This is probably an impossible task for such large models (Beck, 1987, 1997; Stigter *et al.*, 1997). On the other hand, the error in model output, as presented in the previous subsection, is acceptable for speculation and gross conclusions on the functioning of the ecosystem. However, the unspecified prediction error will probably be too large to distinguish between several realistic alternative courses of action.

In the paper conclusions were drawn on the influence of change in the sedimentation of organic matter on the release of phosphorus from the sediments, the effect of the increased nitrate loading on denitrification and sediment phosphorus release, the switch from nitrogen limited to phosphorus limited growth, the competition between cyanobacteria and green algae, the effect of a changed phytoplankton composition on transparency and the effect of increased grazing. How 'hard' are these conclusions? Some conclusions were confirmed by measurements, for example the relationship between the nitrate concentration and denitrification (Van Luijn, 1997). Other conclusions are only hypotheses. The result that the primary production of phytoplankton sustains a lower zooplankton biomass than imposed allows speculation on the limitations of combined strategies of bottom-up control by nutrient limited phytoplankton growth and top-down control by increased phytoplankton losses due to grazing. The simulation of the effect of increased zooplankton grazing may serve as an example of 'educated speculation', because the grazing module has not been tested properly for field conditions. We are faced with the dilemma, similar to the one pointed out in paragraph 3.2, that a relatively complete model gives imprecise predictions, because 'completeness' introduces uncertainties. Further improvement of the credibility of these conclusions and hypotheses may be achieved by specific field and laboratory experiments and by applying specific modules with more restricted boundaries. For example, measured nutrient concentrations may be input in a phytoplankton module to give more precise conclusions about phytoplankton growth limitation.

2a, 2b motivation and constraints

An important motivation to develop large deterministic eutrophication models was to be able to address a variety of questions from managers, both on a regional and a national scale. More specifically, the motivation for this study was to increase knowledge on the lake functioning and to illustrate the possibilities of the model with respect to this.

DBS evolved from predecessors in a couple of years. The costs of model development and initial testing were about 200,000 ECU. The model structure is more or less fixed and covers a wide range of water quality variables. Alterations in schematisation and parameter setting make it possible to adjust the model to specific systems. Constraints in time and money do not often permit the development of a new, better designed model (see also paragraph 4.2). Nevertheless, adjustments and extensions are continuously made to the structure of the model, to be able to meet 'new' questions of managers. With respect to this, the flexibility of DBS was recently improved by reorganising the model into several modules and adding a library with a variety of process formulations.

Turn-over of models and objectives

The development of DBS and its predecessors illustrate that the time-scale of the development of a complex model seems to be larger than the life cycle of actual management questions. Once the nutrient - chlorophyll-a model became established, questions were directed towards algal composition. After inclusion of several phytoplankton species, transparency became an issue. Transparency may be increased following a bottom-up approach (lowering the phytoplankton biomass by means of reducing nutrient availability), but also following a top-down approach (stimulation of grazing pressure on phytoplankton). Therefore, zooplankton came to the fore and later also the freshwater mussel *Dreissena polymorpha*. Nowadays ecological restoration is an important issue, pointing to macrophytes, macrofauna, fish, birds and even mammals as potential model variables. For some of these variables new information is needed, e.g. toxic stress and recreational disruption.

2c, 2d arguments and consequences

Arguments for the approval of the model (results) are not specified in the paper. The model has a sound reputation, because it was frequently applied on a local and on a regional scale by Delft

Hydraulics and several lake managers, and on a national scale under the authority of the Institute for Inland Water Management and Waste Water Treatment. However, acceptability cannot be based on credibility as no serious uncertainty analysis was carried out.

The model application increased understanding of the lake ecosystem, hypotheses were formulated and new research topics were identified. Furthermore, the recommendations based on the study may be qualified as 'soft' instead of 'absolute'. It is argued in the paper reviewed that additional measures to decrease the internal load of ortho-phosphate are not promising, while a further decrease of the external phosphorus load will be effective. Also, flushing was found to be less effective than previously. Optimisation of this measure was suggested by restricting flushing to periods with relatively high phosphorus concentrations in the other inlet water. Finally, increased grazing on phytoplankton turned out to be beneficial for the water transparency, but the feasibility of the enforced grazing was questioned. These recommendations contributed to the actual management strategy, as no further measures for the sediment were carried out and flushing was directed partly to a neighbouring lake. At the moment Lake Veluwe is clear and partially covered with macrophytes (Noordhuis, 1997; Van der Molen *et al.*, 1998b; Van den Berg *et al.*, 1998).

4.2 Acceptability in practice

In the previous paragraphs three model applications were reviewed using the criteria for credibility and acceptability from paragraph 3.4. The discussion on credibility received ample treatment compared with acceptability. This is mainly because the reviewer is a modeller rather than a manager. Therefore, in the following paragraphs acceptability will be highlighted from the point of view of the managers. 'Managers' form a divers group. In Figure 7 (operational) managers are distinguished from policy makers (or decision makers). The information in the following paragraphs has been derived from the level of the operational management, as they are mostly involved with modelling.

Three interviews were carried out with operational lake managers of three districts in The Netherlands. In each interview one specific project was taken as the main subject, but some aspects of acceptability were illustrated with other projects as well. The interviews were structured using a checklist that was reviewed in advance by a communication expert. However, the small number of interviews allowed a qualitative interpretation of the information only.

During the interviews emphasis was put on the initiation of the projects and on the acceptance and implementation of the results. Subjects addressed were the motivation to initiate a modelling project, the constraints in time and money, the arguments for the acceptation or rejection of the model (results), and finally the impact of the results of the modelling project. These items are closely related to the criteria for acceptability (paragraph 3.4). According to the framework of this thesis, modellers address uncertainties and managers accept or reject this part of the study. However, in practice there was an interesting interrelation between manager and modeller with respect to the handling of uncertainties. Managers controlled the effort that modellers may put in assessing uncertainties, modellers frequently involved managers in decisions affecting uncertainties in the beginning of the project and, finally managers interpreted uncertainties (see also Rowe, 1992). Therefore, in an additional paragraph (4.2.3) the handling of uncertainties in the projects reviewed has been addressed.

4.2.1 Motivation

The context for the initiation of modelling projects varies. Morgan & Henrion (1990) distinguished four classes of motivations. Under the first, *substance-focused* motivations, the interest is the substance of the problem itself. The goal is to increase understanding or to obtain an answer to specifically formulated questions and alternatives. The results of a study are also important in analyses undertaken with *position-focused* motivations. Then only a subset of the results is used to provide arguments for one's view or to justify an action (to be) taken. Furthermore, a study may be performed with a *process-focused* motivation. In this case the results of a study are essentially irrelevant as long as they do not become inauspicious. The real motivation is to persuade others that things are under control, or just because it is prescribed by the law or other agreements. Reiterative policy analyses in The Netherlands are partly based on this motivation. Finally, *analyst-focused* motivations may be distinguished. Analysts may derive professional recognition or simply enjoyment from a study, or a specific problem may be used as an example only to develop, test, or demonstrate a new model. The study of Van der Molen & Boers (1994) is basically substance-focused, but it contains also elements of this analyst-focused motivation. Of course, in practice usually a combination of motivations will occur.

The operational managers interviewed described the motivation of their projects mostly as substance-focused. In some cases emphasis was put on the results of different alternatives (e.g. location of effluent discharges, efficiency of lake restoration measures), in others increasing understanding of the aquatic ecosystem was the major motivation. In one case it was a priori assumed that 'something had to be done', so the motivation was partly position-focused. In most cases the operational managers initiated the study, driven by general goals in agreement with the higher administrative levels of the management. Since the nineties the use of models has become less obvious for the policy makers. This may be due to disappointing experiences in the past, but it was also argued that 'models are not needed as there are experts in the institute'. In one case a research institute proposed the project to the water manager with an underlying objective to increase the public support for their institute, thereby trying to influence the discussion about reorganisation and especially relocation of the institute. This is a kind of process-focused motivation.

4.2.2 Constraints

Constraints in time and money were more strict when models were used to calculate alternatives when compared with motivations to increase knowledge on the system. Furthermore, constraints were generally more inflexible when the manager also obtained money from another contracting party. Apart from the motivations to initiate a study, especially constraints in time and money were important for the choice of the *model (formulations)*, the *variables* taken into account and the attention paid to *systems analysis and uncertainties.*

model (formulations)

In one project a new model was developed as existing models were not available or considered to be inappropriate. In another project an existing model was adjusted significantly to make it appropriate for the specific water system. In other projects existing models were applied or only slightly altered, as it was argued that there were no resources to set up a new model. The development of the deterministic models used in these projects was indeed time-consuming (several years). Even systems analysis based on an existing model may take more than one year.

However, the managers' choice of the model and the modeller was affected by other factors than suitability and constraints. Informal and personal relationships and accidental factors contributed to this choice. The reputation of existing models, of their developers/users and of the institutes to which they belonged, played a role. Furthermore, in most cases there was already a kind of alliance of the managers with the developers/users, because of preceding projects (resulting in system specific knowledge or even availability of applied water quantity models), foregoing jobs of managers at the modelling institute or vice versa. Finally, geographical coincidence of the locations of the institutes of both representatives was found to be of importance. These factors affected the relation between operational manager and modeller on one hand and between operational manager and higher levels in the management on the other.

variables

In one project, in which a new model was built, only phosphorus was studied. Nitrogen was dismissed in the beginning and phytoplankton at the end of the project as a consequence of constraints in time and money. Consequently, the initial objectives (effects of flushing, nutrient emissions from polders and sediments on eutrophication in the lakes) remained partly unanswered. In case an existing model was adopted, managers conformed with the available variables. Therefore, the translation of the problem into objectives was restricted by the model.

systems analysis and uncertainties

In the projects the system analytical approach was followed in various ways. In one project a system analytical approach in accordance with the methodology described in chapter 2 was chosen from the beginning (Brinkman *et al.*, 1989b), but not all stages were addressed sufficiently. In the other projects one or more existing models were applied. In one of these projects an existing model was adjusted by modellers. The schematisation and further stages of systems analysis are currently being performed by operational managers. In all projects, constraints in time and money were the main reason for not following systems analysis entirely and for the incomplete assessment of uncertainties.

Analysis of uncertainties was mostly not an objective for managers and priority was given to for example the calculation of alternatives. In only one project managers imposed constraints also with respect to the precision of the results. However, the modellers refused to apply their model as the constraints were found to be unattainable. On the other hand, in the projects reviewed modellers never refused a project because of the low priority on uncertainty analysis. Afterwards, insufficient addressing of uncertainties was sometimes admitted and recommendations were made for further model analyses (Brinkman *et al.*, 1989a, b; Van der Molen, 1994).

4.2.3 Handling of uncertainties

There are several possible explanations why modellers and managers disregard uncertainties, and there are several incentives for addressing, managing, or at least being aware of uncertainties (e.g. Funtowicz & Ravetz, 1990; Barkman, 1997). They are categorised in *methodological, strategic* and *communicative.*

methodological

Taking into account uncertainties is laborious and complex. The assessment of uncertainties in environmental models can neither be complete nor free from value-judgement. In environmental models it is not possible to account for all uncertainties in input data, parameters, model formulations and observations due to incompleteness of the available information. The choice of methods, weight factors, etc. is subjective, depending on the experience of the modellers, their interpretation of the objectives and even on hard and software facilities. However, a systematic evaluation of uncertainties during model development may facilitate the proper choice of variables, parameters and processes in the model. During application it may guide further research and it may serve as a cost-effective resource allocation for reducing uncertainties in input data. Furthermore, a systematic management of uncertainties may assist in the development of new strategies by identifying shortcomings in the previous ones.

strategic

Both modellers and managers may be embarrassed and reluctant to address the presence and magnitude of uncertainties in their work. Furthermore, the explicit introduction of uncertainty may provide certain actors with possibilities of proposing a delay of action due to inadequate scientific information. The assessment of uncertainties may limit the authority of scientific knowledge in the political process and therefore the model (results) may be presented as 'hard'. On the other hand, an explicit assessment of uncertainties will produce better defensible results as their quality is better quantified. The results are also more flexible, because small variations in the input conditions may be interpreted easier without new model calculations. Furthermore, both modellers and managers have the responsibility to provide users of their products with information on the implications and limitations of their work. In the end this will contribute to a better understanding of the credibility of the model(ler) and therefore contribute to the confidence in the authority of modeller and manager.

communicative

Finally, uncertainties are not easy to communicate. Not everybody is trained to interpret statistical properties of model results, and managers may prefer 'yes/no' or 'go/no go' answers. However, communication may be more effective by giving appropriate weight to the model results.

Experiences of two projects at the same institute and with very similar models showed that operational managers convinced policy makers easier of the necessity of a project when the model was presented as all-embracing instead of 'a technical aid inflicted with uncertainties'. There is a temporal (learning) aspect in this as the lack of awareness of managers for uncertainty was found especially in the earlier projects of the eighties. Since the nineties, managers have been more aware of (large) uncertainties in models and their outcomes, even if they are not quantified. Next, the handling of uncertainties in the projects reviewed will be illustrated for the stages: set-up of a conceptual model, sensitivity analysis, parameter estimation, validation and finally, the use of the results.

In the projects reviewed, modellers and managers discussed the available alternatives concerning the model variables, processes and the schematisation. For example, in all cases top-down interactions were omitted or included as simple input functions on algal biomass, because the uncertainties in the data on these specific processes, and therefore in model parameters and formulations, were considered to be too high. In some projects this was even stated explicitly, but it was never tested properly. Another example is the choice whether or not algal species should be included or what algal species should be included. Accounting for uncertainties with respect to the schematisation may be illustrated by the decision in one project to study a part of the water system in detail and to extrapolate the results to the total area, instead of modelling the complete water system, in order to decrease uncertainties. Furthermore, measures were expected to be most effective in this specific part of the water system (Brinkman *et al.*, 1989a). However, this assumption was based on expert knowledge only and not properly tested. In practice, the open boundaries of the study area implied large uncertainties. These would have been reduced if the complete water system had been modelled, but in that case the modellers would have been faced, for example, with modelling a wide variation in soil types.

In the project, in which a new model was developed, a sensitivity analysis was performed after parameter estimation by comparing the results of the two scenarios with the most extreme results using another set of parameters. The available data were used for parameter estimation, so no validation on an independent set of data could be performed. The parameter estimation was supported with a residue-analysis. Despite some periodicity in the residues, it was concluded that it was not necessary to modify the model (Brinkman *et al.*, 1989b). In the project where an existing model has been modified significantly a detailed sensitivity analysis of inputs and parameters is being carried out at this moment to consider the feasibility of discriminating between alternative courses of action. Normally, in the case existing models were applied sensitivity analysis and parameter estimation were carried out in previous, independent projects (see paragraph 4.1.3). Additional parameter estimation for the specific project may be seen as fine tuning. In one project all parameters but one were found to obtain the same value as in a previous parameter estimation

Decision support systems

In addition to excuses not to stress uncertainties, several operational managers expressed their opinion upon 'large' decision support systems. According to them, these systems give rise to the false suggestion that all aspects are included in the study and hamper further considerations. In one case these tools were used at the level of operational management, but only some results were passed on to the policy makers. Furthermore, the loss of detail by using decision support systems was mentioned. The operational managers are afraid to lose control of the process. Of course these arguments are in contradiction with the intention of decision support aids (e.g. Van Rooy, 1997).

procedure for a neighbouring lake (Van der Molen, 1994). In two projects the results of the parameter estimation were validated visually on data of another period of the same lake.

Parameter estimation and validation were performed only visually and the parameters that were varied or their ranges were even not always mentioned. Correlation between parameters, and therefore the possibility of different combinations of parameter values resulting in comparable outcomes was not addressed. In this stage of the project qualitative remarks about uncertainties in input data and observations were sometimes made.

As pointed out in paragraph 4.2.2, uncertainties were seldom an explicit issue for the operational water managers. In all cases managers imposed modellers with constraints that allowed an incomplete handling of uncertainties only. Apart from this, also strategic and communicative arguments were mentioned by lake managers as explanation for the low priority for uncertainties. In a recent project operational managers stressed probably insignificant small differences between alternatives by expressing the results in percentages to make the alternatives better distinguishable. Moreover, addressing uncertainties was considered to be 'not functional' and 'confusing'. The use of probabilities and ranges was even disapproved as 'suggesting a high accuracy'.

In conclusion, although implicitly several aspects of uncertainties were addressed, a great deal can still be improved. Managers may be more willing to facilitate modellers to assess uncertainties in a proper way, when they become more aware of the benefits of this extension to their projects.

4.2.4 Acceptance or rejection?

Given the constraints in time and money and the resulting incomplete assessment of uncertainties as mentioned in the previous paragraphs, it is not surprising that the propagation of assumptions and uncertainties in the predictions was not addressed systematically. One project, which is not finished yet, may become the proverbial exception because a start was made with a proper handling of the various stages of systems analysis.

In one project reviewed, the results were rejected after validation by the managers. In all other projects, the results were accepted, although arguments were mostly vague or even lacking. Instead results of systems analysis were sometimes summarised briefly: 'the model gives a sufficient accurate picture to legitimate global conclusions about imperative measures' (Brinkman *et al.*, 1989b).

Frustrations of modellers

There are numerous benefits to modellers if they are able to work in close operation with (lake) managers instead of executing an isolated study. Managers have knowledge of the water system involved and may inspire the modeller with realistic objectives and alternatives.

However, there are also disadvantages. Managers admitted that they tend to promise more data and data of better quality, than they can actually supply. Furthermore, the delivery of the data is mostly later than promised. Also, managers sometimes adjust objectives (alternatives) during the study (see also paragraph 4.1.3), without judging the study as a methodological approach to refine objectives and guide further research (analogously to paragraph 4.2.3).

In other projects modellers and managers were generally 'satisfied' with respect to the results. Approval of the results always applied to the complete project; there were no conditions for specific parts of the results (e.g. Reeders & Helmerhorst, 1996).

The (lack of) arguments of managers for acceptance or rejection were affected by their motivation to initiate the study (paragraph 4.2.1) and the constraints imposed by themselves (paragraph 4.2.2). The constraints determined to a large extent the weight that is put on credibility. Furthermore, the relation between manager and modeller was important, and also the authority of the modelling institution was mentioned frequently as a guarantee for a 'good modelling practice' (paragraph 4.2.2).

4.2.5 Impact of the projects

The rather simplified interrelation between modeller and manager, as illustrated in Figure 7 in paragraph 3.3, was to a large extent consistent with the projects reviewed. Model results were first judged by operational managers, next they extracted information from the study and, finally, they supplied this to managers at the policy making level. Then several other actors became involved before implementation of the model results. Accordingly, the impact of a project depended not only on the quality of the study but also on how managers fitted the study in their decision making (Morgan, 1978; Van Rooy, 1997).

In one project the results were presented in a symposium, passed on to the administrative level and incorporated in other documents (e.g. Provincie Friesland, 1989). However, the operational managers now believe that the study had little impact. Credibility of the modelling study was not a matter of debate and did not affect the impact according to the operational management. More positively, the policy makers concluded that the models can be used to increase understanding of the effect of measures, but not how to realise them (Brinkman *et al.*, 1989a). The fact that the results of the study were not extrapolated to the complete water system and the non-feasibility of the result-oriented alternatives may have played a role in the low impact of the results. Furthermore, the modelling study was performed independently from decision making; during the project the operational managers did not explicitly undertake actions to increase applicability of the results. The transfer of the model from modellers to managers was a side-objective only. According to the management this failed because of lack of transparency of the model and lack of continuity in their relation with the developers. However, the consequences of the use of models were underestimated as user-friendliness was not specified in advance and employees were not trained in working with models.

In another project model results were copied unabridged in documents for decision making and measures are in preparation or in realisation (Reeders & Helmerhorst, 1996; Reeders, 1997; T.H. Helmerhorst, personal communication). Compared with the previous project, the organisation was better equipped for decision-making: the operational managers participated also at the decision making level and informed other actors during the modelling project. Credibility of the model results was no major issue. One actor, who had to incur high cost for the realisation of alternatives, initiated a 'second opinion' study, but this had no impact on the decisions.

Knowledge derived from model studies was also used for improving advise and decisions on variables that were not covered by the model. This may be seen as a kind of 'educated speculation' (paragraph 3.2). However, even having access to a model was put forward as increasing the authority of the managers.

5 Synthesis

In this thesis the credibility of eutrophication models and their relevance to water quality management are studied. Therefore, a system analytical approach is presented and illustrated with examples from the field of eutrophication of freshwater lakes. Contrary to the usual practice of systems analysis, credibility - defined as the technical appropriateness of the model and its results - is explicitly distinguished from acceptability - defined as the users' perception of the practical value of the model and its results. This separation reflects actual practice; it is in agreement with the examples described in chapter 4, i.e. the papers reviewed and the case studies evaluated using interviews with lake managers. The separation of credibility and acceptability facilitates the discussion of the role of eutrophication models in water management, and plays therefore a leading role in this thesis.

Both credibility and acceptability are specified in a set of criteria with the purpose to serve as a checklist for both modellers and managers. The criteria proved to fulfil a purpose in reviewing the papers and evaluating a number of projects on modelling eutrophication. The assignment of credibility is assumed to be the responsibility of the modeller, while acceptability is related to the users or the managers. In the examples of chapter 4 'managers' are restricted to managers of the operational level, but some of the arguments may concern managers from the policy making level as well.

Modellers may produce a credible model or credible model results by carefully pursuing systems analysis. Uncertainties pop up in all stages of this approach. Methodological incentives are the main reason why uncertainties receive little attention in practice. It is a laborious, complex job, there is no uniform procedure and it is not possible to be complete and free of value-judgement when environmental models are concerned. Nevertheless, credibility of a model and of model results necessitates accounting for uncertainties. It is advocated here that the recognition of uncertainties should be made explicit, even if only little attention is paid to it. This makes it possible to make a fair consideration when acceptability comes into sight.

After the assessment of credibility by modellers, managers have to accept or reject (parts of) the model or its results. In practice, acceptability of a model or model results depends not only on credibility of the modelling exercise. Before the actual use of a model, informal and personal relationships and accidental factors appear to contribute significantly to the choice of a manager for a model(ler). Acceptability is further affected by the motivations to undertake a project and by the constraints in time and money imposed by managers. They, in their turn, are willing to skip part of the objectives, to adjust their problems to the possibilities of existing models and they pay little attention to uncertainties involved with modelling.

Water managers use eutrophication models mostly to gain insight in the water system. This insight is further utilised for decision making and for the initiation of new research. Several decisions were made without specification of uncertainties and not seldom they have not been missed afterwards either. Models are also used to calculate alternative courses of action, but results were mostly interpreted qualitatively instead of absolutely. In this way the large uncertainties involved with eutrophication modelling are implicitly and partly accounted for. Constraints in time and money imposed by managers are often the reason behind the methodological barrier for the modeller to address uncertainties sufficiently. However, managers may take advantage from the assessment of uncertainties. It may guide further research, assist in the development of new strategies, save time and money in certain situations, enhance public support and contribute to the confidence in the authority. Moreover, managers may try to be more explicit in their arguments to accept or reject (parts of) the model or its results, analogously to the modeller with respect to credibility. This may reveal the shortcomings in present eutrophication modelling projects and may yield more transparent information in policy making. This, in turn, may enlarge and improve the role of eutrophication models in water management.

References

Barkman, A., 1997. Applying the critical loads concept: constraints induced by data uncertainty. Reports in ecology and environmental engineering; report 1. Department of Chemical Engineering II, Lund University, Sweden.

Beck, M.B., 1981. Hard or soft environmental systems? *Ecological Modelling* 11: 233-251.

Beck, M.B., 1983. Uncertainty, system identification, and the prediction of water quality. In: M.B. Beck & G. Van Straten (eds.), Uncertainty and Forecasting of Water Quality. Springer-Verlag, Berlin, pp. 3-68.

Beck, M.B., 1987. Water quality modeling: a review of the analysis of uncertainty. *Water Resources Research* 23: 1393-1442.

Beck, M.B., 1997. Applying systems analysis in managing the water environment: towards a new agenda. *Water Science and Technology* 36, 5: 1-17.

Benndorf, J., 1987. Food web manipulation without nutrient control: a useful strategy in lake restoration? *Schweizerische Zeitschrift für Hydrologie* 49/2: 237-248.

Beven, K.J., 1993. Prophecy, reality and uncertainty in distributed hydrological modelling. *Advances in Water Resources* 16: 41-51.

Biswas, A.K., 1976. Systems approach to water mangement. MacGraw-Hill, New York.

Boekholt, A.H., 1996a. Interlaborariumonderzoek. Project 105 - algemene parameters + metalen in oppervlaktewater - 16 april 1996. RIZA werkdocument 96.109X, Lelystad (in Dutch).

Boekholt, A.H., 1996b. Interlaborariumonderzoek. Project 108 - BOD, COD en nutriënten in oppervlaktewater - 16 september 1996. RIZA werkdocument 96.112X, Lelystad (in Dutch).

Boekholt, A.H., 1996c. Interlaborariumonderzoek. Project 110 - chlorophyll - 16 september 1996. RIZA werkdocument 96.114X (in Dutch).

Boekholt, A.H., 1996d. Interlaborariumonderzoek. Project 115 - oppervlaktewater, metalen + algemene chemische parameters - 11 november 1996. RIZA werkdocument 96.119X, Lelystad (in Dutch).

Boekholt, A.H., 1997a. Interlaborariumonderzoek. Project 126 - algemene parameters en metalen in oppervlaktewater - 9 juni 1996. RIZA werkdocument 97. 072X, Lelystad (in Dutch).

Boekholt, A.H., 1997b. Interlaborariumonderzoek. Project 128 - chlorophyll-a in oppervlaktewater - 8 september 1997. RIZA werkdocument 97.074X, Lelystad (in Dutch).

Boekholt, A.H., 1998a. Interlaborariumonderzoek. Project 133 - algemene parameters en metalen in oppervlaktewater - 20 oktober 1997. RIZA werkdocument 97. 079X, Lelystad (in Dutch).

Boekholt, A.H., 1998b. Interlaborariumonderzoek. Project 144 - algemene parameters en metalen in oppervlaktewater - 25 mei 1998. RIZA werkdocument 98. 061X, Lelystad (in Dutch).

Boekholt, A.H., 1998c. Interlaborariumonderzoek. Project 147 - chlorophyll-a in oppervlaktewater - 6 juli 1998. RIZA werkdocument 98.062X, Lelystad (in Dutch).

Boers, P.C.M. 1996. Nutrient emissions from agriculture in The Netherlands, causes and remedies. *Water Science and Technology* 33, 4-5: 183-189.

Boers, P.C.M., P.A. Finke, J.J.M. Van Grinsven & P. Groenendijk, 1996. Definitie en haalbaarheid van een interdepartementaal model voor de berekening van de nutriëntenemissie naar grondwater en oppervlaktewater. RIZA werdocument 96.081X, Lelystad (in Dutch).

Boers, P.C.M., L. Van Ballegooijen & E.J.B. Uunk, 1991. Changes in phosphorus cycling in a shallow lake due to food web manipulations. *Freshwater Biology* 25: 9-20.

Boers, P.C.M. & D.T. Van der Molen, 1992. Invloed van fosfaatnalevering op de fosfaatconcentratie in meren voor en na sanering van de externe belasting. RIZA nota 92.021, Lelystad (in Dutch).

Boers, P.C.M., I.H.S. Hoek, F. Klijn, M.A. Menke, A,F.M. Meuleman & D.T. Van der Molen (red.), 1995. 'Vermesting' van oppervlaktewater. *Landschap* 6 (in Dutch with English summaries).

Boström, B., M. Jansson & C. Forsberg, 1982. Phosphorus release from lake sediments. *Archiv für Hydrobiologie* 18: 5-59.

Brinkman, A.G., H.J.W.J. Van Huet, 1989a. Fosfaatonderzoek Fosfri. Verslag van veld-, laboratorium en modelonderzoek in het zuidwest Friese merengebied gedurende de periode 1984-1989. Universiteit Twente, Enschede; Limnologisch Instituut, Oosterzee (in Dutch).

Brinkman, A.G. ., H.J.W.J. Van Huet, 1989b. Fosfaatonderzoek Fosfri. Verslag van veld-, laboratorium en modelonderzoek in het zuidwest Friese merengebied gedurende de periode 1984-1989. Bijlagen. Universiteit Twente, Enschede; Limnologisch Instituut, Oosterzee (in Dutch).

Brinkman, A.G. & W. Van Raaphorst, 1986. De fosfaathuishouding in het Veluwemeer. Proefschrift Technische Hogeschool Twente, The Netherlands (in Dutch with an English summary).

Canale, R. (ed.), 1976. Modeling Biochemical Processes in Aquatic Ecosystems. Ann Arbor Science Publishers, Michigan USA.

Clarke, R.T., 1973. A review of some mathematical models used in hydrology, with observations on their calibration and use. *Journal of Hydrology* 19: 1-20.

Cooke, G.D., E.B. Welch, S.A. Peterson & P.R. Newroth, 1993. Restoration and management of lakes and reservoirs. Lewis Publishers, Boca Raton, FL.

CUWVO, 1976. Schets van de eutrofiëringssituatie van het Nederlandse oppervlaktewater en overzicht van de ondrzoeksactiviteiten van de verschillende waterkwaliteitsbeheerders. Resultaten van de eerste eutrofiëringsenquête. Coördinatiecommissie Uitvoerinig Wet Verontreiniging Oppervlaktewateren, Werkgroep VI (in Dutch).

CUWVO, 1980. Ontwikkeling van grenswaarden voor doorzicht, chlorophyl, fosfaat en stikstof. Resultaten van de tweede eutrofiëringsenquête. Coördinatiecommissie Uitvoerinig Wet Verontreiniging Oppervlaktewateren. RIZA, Lelystad (in Dutch).

CUWVO, 1987. Vergelijkend onderzoek naar de eutrofiëring in nederlandse meren en plassen. Resultaten van de derde eutrofiëringsenquête. Coördinatiecommissie Uitvoerinig Wet Verontreiniging Oppervlaktewateren, Werkgroep VI (in Dutch).

Danen-Louwerse, H.J., R. Portielje & L. Lijklema, 1996. Phosphorus fixation in lake sediments. The case of Lake Veluwe. Agricultural University Wageningen, Department of Water Quality Management and Aquatic Ecology, report M274.

De Rooij, N.M., 1991. HADES; Ontwikkeling en verkennende berekeningen. T584, Waterloopkundig Laboratorium, The Netherlands (in Dutch).

Delft Hydraulics, 1994. DBS model coefficients; results of a model validation study for DBS in networks. T1210, Delft Hydraulics, The Netherlands.

Droop, M.R., 1973. Some thoughts on nutrient limitation in algae. *Journal of Phycology* 9: 264-272.

Eykhoff, P., 1974. System identification. Parameter and state estimation. John Wiley & Sons, New York.

Funtowicz, S. & R. Ravetz, 1990. Uncertainty and quality in science for policy. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Golterman, H.L., 1970. Mogelijke gevolgen van de fosfaat-eutrofiëring van het oppervlaktewater. *H_2O* 3: 209-215 (in Dutch).

Golterman, H.L., 1973. De invloed van fosfaat op het aquatisch milieu. *H_2O* 6: 430-438 (in Dutch).

Goodall, D.W., 1972. Building and testing ecosystem models. In: J.N.J. Jeffers (ed.), Mathematical Models in Ecology. Blackwell, Oxford, UK, pp. 173-194.

Gorham, P.R., 1964. Toxic algae. In: D.F. Jackson (ed.), Algae and Man. New York, Plenum Press, pp. 307-336.

Gulati, R.D., E.H.R.R. Lammens, M.-L. Meijer & E. Van Donk (eds.), 1990. Biomanipulation, tool for water management. Proceedings of an international symposium held in Amsterdam, The Netherlands, 8-11 August, 1989. *Hydrobiologia 200/201*. Kluwer Academic Publishers.

Hieltjes, A.H.M. & L. Lijklema, 1981. Nalevering van fosfaat door sedimenten. V: fosfaat-balans model voor het Brielse Meer. *H_2O* 14: 137-140 (in Dutch).

Hosper, S.H., 1978. De ontwikkeling van richtlijnen voor fosfaatconcentratie en fosfaatbelasting voor Nederlandse wateren. *H_2O* 11: 329-334 (in Dutch).

Hosper, S.H., 1984. Restoration of Lake Veluwe, The Netherlands, by reduction of phosphorus loading and flushing. *Water Science and Technology* 17: 757-768.

Huisman, J., 1997. The struggle for light. Thesis University of Groningen, The Netherlands.

Hutchinson, G.E., 1957. A treatise on limnology. Volume I, geography, physics and chemistry. Chapter 12: The phosphorus cycle in lakes. Wiley and Sons, New York, 727-752.

Ibelings, B.W., 1992. Cyanobacterial waterblooms: the role of buoyancy in water columns of varying stability. Thesis University of Amsterdam, The Netherlands.

Jagtman, E., D.T. Van der Molen & S. Vermij, 1992. The influence of flushing on nutrient dynamics, composition and densities of algae and transparency in Veluwemeer, The Netherlands. *Hydrobiologia* 233: 187 - 196.

Jeppesen, E., P. Kristensen, J.P. Jensen, M. Søndergaard, E. Mortensen & T. Lauridsen, 1991. Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Memorie dell 'Istituto Italiano di Idrobiologie* 48: 127-148.

Jørgensen, S.E. (ed.), 1983. Application of Ecological Modeling in Environmental Management. Elsevier Scientific Publishing Company, Amsterdam.

Jørgensen, S.E., 1988. Use of models as experimental tool to show that structural changes are accompanied by increased exergy. *Ecological Modelling* 41: 117-126.

Jørgensen, S.E., 1992. The shift in species composition and ecological modeling in hydrobiology. *Hydrobiologia* 239: 115-129.

Jørgensen, S.E., 1994. Fundamentals of Ecological Modeling (2nd Edition). *Developments in Environmental Modeling* 19. Elsevier Amsterdam.

Kleeberg, A. & H.-P. Kozerski, 1997. Phosphorus release in Lake Gro(er M(ggelsee and its implications for lake restoration. *Hydrobiologia* 342/343: 9-26.

Lijklema, L., 1982. A dynamic phosphate budget model for a eutrophic lake. *Hydrobiologia* 91, 227-233.

Lijklema, L., 1983. Internal loading. *Water Supply* 1: 35-42.

Lijklema, L., 1986. Phosphorus accumulation in sediments and internal loading. *Hydrobiological Bulletin* 20: 213-224.

Los, F.J., 1980. An algal bloom model as a tool to simulate management measures. In: J. Barica & L.R. Mur (eds.), *Developements in Hydrobiology* 2: 171-178. W. Junk BV Publishers, The Hague.

Los, F.J. & J.J. Brinkman, 1988. Phytoplankton modeling by means of optimization: a 10-year experience with BLOOM II. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 23: 790-795.

Los, F.J., J.C. Stans & N.M. De Rooij, 1988. Eutrofiëringsmodellering van de randmeren. T100, Waterloopkundig Laboratorium Delft, The Netherlands (in Dutch).

Los, F.J., M.T. Villars & M.R.L. Ouboter, 1994. Model validation study; DBS in netwetoks. T1210, Delft Hydraulics, The Netherlands.

Mankin, J.B., R.V. O'Neill, H.H. Shugart & B.W. Rust, 1977. The importance of validation in ecosystem analysis. In: G.S. Innis (ed.), New Directions in the Analysis of Ecological Systems. *Simulation Council Proceedings Series* 5: 63-72.

Meijer, M.-L, E.H. Van Nes, E.H.R.R. Lammens, R.D. Gulati, M.P. Grimm, J. Backx, P. Hollebeek, E.M. Blaauw & A.W. Breukelaar, 1994. The consequences of a drastic fish stock reduction in the large and shallow Lake Wolderwijd, The Netherlands. Can we understand what happened? *Hydrobiologia* 275/276: 31-42.

Ministerie van Verkeer en Waterstaat, 1989. Eutrofiëring. Water voor nu en later; basisrapport derde Nota waterhuishouding. RIZA nota 89.034 (in Dutch with an English summary).

Mooij, W.M., 1990. Een graasmodule voor het eutrofiëringsmodel JSBACH. T642, Waterloopkundig Laboratorium, The Netherlands (in Dutch).

Morgan, M.G., 1978. Bad science and good policy analysis. *Science* 201, 4360.

Morgan, M.G. & M. Henrion, 1990. Uncertainty; A guide dealing with uncertainty in quantitative risk and policy analysis. Cambridge University press, Cambridge, UK.

Noordhuis, R., 1997. Biologische monitoring zoete rijkswateren: watersysteemrapportage Randmeren. RIZA rapport 95.003, Lelystad (in Dutch with an English summary).

O'Neill, R.V. & B. Rust, 1979. Aggregation error in ecological models. *Ecological Modelling* 7: 91-105.

Oreskes, N., K. Shrader-Frechette & K. Belitz, 1994. Verification, validation, and confirmation of numerical models in the earth sciences. *Science* 263: 641-646.

Orleans, A.B.M., F.L.T. Mugge, T. Van der Meij, P. Vos & W.J. Ter Keurs, 1994. Minder nutriënten in het oppervlaktewater door bufferstroken? Een literatuuranalyse. Rapport Milieubiologie RU Leiden (in Dutch).

Patten, B.C. (ed.), 1971-1976. Systems Analysis and Simulation in Ecology. Vols.1-4. Academic Press, New York.

PER, 1982. Eutrofiëringsonderzoek Wolderwijd-Nuldernauw. Projectgroep Eutrofiërings-onderzoek Randmeren, Rijkswaterstaat, Lelystad (in Dutch).

PER, 1986. Bestrijding van de eutrofiëring van het Veluwemeer-Drontermeer. Projectgroep Eutrofiëringsonderzoek Randmeren, Rijkswaterstaat, Lelystad (in Dutch).

Pintér, J., 1990. Globally optimized calibration of environmental models. *Annals of Operations Research* 25: 211-222.

Portielje, R. & D.T. Van der Molen, 1998. Relaties tussen eutrofiëringsvariabelen en systeemkenmerken van de Nederlandse meren en plassen. Deelrapport II voor de Vierde Eutrofiëringsenquête. RIZA rapport 98.007, Lelystad (in Dutch).

Provincie Friesland, 1989. Bestuursopdracht eutrofiëringsbestrijdingsplan. Provincie Friesland, Hoofdgroep Waterstaat en Milieu, Afdeling Waterkwaliteitsbeheer (in Dutch).

Reckhow, K.H., 1979. Empirical lake models for phosphorus: development, applications, limitations, and uncertainty. In: D. Scavia & A. Robertson (eds.), Perspectives on Lake Ecosystem Modeling. Ann Arbor Science Publishers, Michigan USA, pp. 193-222.

Reeders, H.H., 1997. Delta Schuitenbeek: een beleidsvoorbereidende studie. Rapport 97.01, Rijkswaterstaat Directie IJsselmeergebied, Lelystad (in Dutch).

Reeders, H.H., P.C.M. Boers, D.T. Van der Molen & T.H. Helmerhorst, 1998. Cyanobacterial dominance in the lakes Veluwemeer and Wolderwijd, The Netherlands. *Water Science and Technology* 37, 3: 85-92.

Reeders, H.H. & T.H. Helmerhorst, 1996. Op weg naar helderheid. Een heroriëntatie van BOVAR gericht op 2000. Rapport 96.01, Rijkswaterstaat Directie IJsselmeergebied, Lelystad (in Dutch).

Reynolds, C.S., 1987. The response of phytoplankton communities to changing lake environments. *Schweizerische Zeitschrift für Hydrologie* 49: 220-236.

Rijsdijk, R.E., 1995. Empirische relaties en deterministische modellen; toepassing in nederlandse eutrofe plassen en meren. RIZA werkdocument 95.025X, Lelystad (in Dutch).

Rowe, W.D., 1992. Risk analysis: a tool for policy decisions. In: M. Waterstone (ed.), Risk and society: the interaction of science, technology and public policy. Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 17-31.

Rowe, W.D., 1994. Understanding uncertainty. *Risk Analysis* 14: 743-750.

Rykiel, E.J.R., 1996. Testing ecological models: the meaning of validation. *Ecological Modelling* 90: 229-244.

Salt, G.W., 1983. Roles: their limits and responsibilities in ecological and evolutionary research. *American Naturalist* 122: 697-705.

Sas, H. (ed.), 1989. Lake restoration by reduction of nutrient loading: expectations, experiences and extrapolations. Academia Verlag Richarz GmbH, St. Augustin, Germany.

Scholten, H. & M.W.M. Van der Tol, 1994. Towards a metrics for simulation model validation. In: J. Grasman & G. Van Straten, Predictability and Nonlinear Modeling in Natural Sciences and Economics. Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 398-410.

Simons, T.J. & D.C.L. Lam, 1980. Some limitations of water quality models for large lakes: a case study of lake Ontario. *Water Resources Research* 16: 105-116.

Smayda, T.J., 1997. What is a bloom? A commentary. *Limnology and Oceanography* 42: 1132-1136.

Smits, J.G.C. & D.T. Van der Molen, 1993. Application of SWITCH, a model for sediment water exchange of nutrients, to Lake Veluwe in The Netherlands. *Hydrobiologia* 253: 281-300.

Stigter, J.D., M.B. Beck & R.J. Gilbert, 1997. Identification of model structure for photosynthesis and respiration of algal populations. *Water Science and Technology* 36, 5: 35-42.

Straskraba, M., 1979. Natural control mechanisms in models of aquatic ecosystems. *Ecological Modelling* 74: 1-38.

Straskraba, M., 1994. Ecotechnological models for reservoir water quality management. *Ecological Modelling* 6: 305-321.

Straskraba, M. & A. Gnauck, 1985. Freshwater Ecosystems; modelling and simulation. *Developments in Environmental Modeling* 8. Elsevier Science Publishers, Amsterdam, The Netherlands.

Thomann, R.V., 1982. Verification of water quality models. *Journal of Environmental Engineering* 108 (EE5): 923-940.

Thomann, R.V. & J.A. Mueller, 1987. Principles of Surface Water Quality Modeling and Control. Harper & Row Publisher, New York.

Van Ballegooijen, L. & D.T. Van der Molen, 1993. De invloed van invoergegevens op het eutrofiëringsmodel DBS: analyse van de modeluitkomsten voor het Veluwemeer voor de periode 1985-1987. RIZA werkdocument 93.078X, Lelystad (in Dutch).

Van Ballegooijen, L. & D.T. Van der Molen, 1994. Water- en nutriëntenbalansen voor de randmeren van de Roggebotsluis tot de Nijkerkersluis over de periode 1976 - 1992. RIZA werkdocument 94.082X, Lelystad (in Dutch).

Van Beek, E., 1993. Duurzame modellering voor duurzaam waterbeheer. Inaugurele rede, Technische Universiteit Delft.

Van den Berg, M.S., H. Coops, M.-L. Meijer, M. Scheffer & J. Simons, 1998. Clear water associated with a dense Chara vegetation in the shallow and turbid lake Veluwemeer. *Ecological Studies* 131: 339-352.

Van der Molen, D.T., 1991. A simple, dynamical model for the simulation of the release of phosphorus from sediments in shallow, eutrophic systems. *Water Research* 25: 737 - 744.

Van der Molen, D.T., 1994. Berekeningen naar de effectiviteit van verschillende afleidings-alternatieven van de Schuitenbeek en het gemaal Putterhuis op de fosfor-concentratiein het Wolderwijd-Nuldernauw. RIZA werkdocument 94.089X, Lelystad (in Dutch).

Van der Molen, D.T. & P.C.M. Boers, 1994. Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. *Hydrobiologia* 275/276: 379-389.

Van der Molen, D.T. & P.C.M. Boers, 1996. Changes in phosphorus and nitrogen cycling following food web manipulations in a shallow Dutch lake. *Freshwater Biology* 35: 189-202.

Van der Molen, D.T., R. Kruijne & E.J.B. Uunk, 1998a. Verwijdering van stikstof en fosfor door bufferstroken langs de Mosbeek. *H_2O* 31, 3: 26-28 (in Dutch).

Van der Molen, D.T., F.J. Los, L. Van Ballegooijen & M.P. Van der Vat, 1994. Mathematical modeling as a tool for management in eutrophication control of shallow lakes. *Hydrobiologia* 275/276: 479-492.

Van der Molen, D.T. & J. Pintér, 1993. Environmental model calibration under different specifications: application to the model SED. *Ecological Modelling* 68: 1-19.

Van der Molen, D.T., R. Portielje, P.C.M. Boers & L. Lijklema, 1998b. Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, The Netherlands. *Water Research* 32: 3281-3288.

Van der Molen, D.T., R. Portielje & S.P. Klapwijk, 1998c. Vierde Eutrofiëringsenquête van de Nederlandse meren en plassen. *H_2O* 31, 11: 16-22 (in Dutch with an English summary).

Van der Molen, D.T., L. Van Ballegooijen & P.C.M. Boers, 1993. Worden de waterkwaliteitsdoelstellingen van het Veluwemeer gehaald? Modelonderzoek naar enkele beheersscenario's. RIZA nota 93.034, Lelystad (in Dutch).

Van der Vat, M., 1995. Extension of the grazing module of the phytoplankton model BLOOM II. T1555, Delft Hydraulics, The Netherlands.

Van Luijn, F., 1997. Nitrogen removal by denitrification in the sediments of a shallow lake. Thesis Landbouwuniversiteit Wageningen, The Netherlands.

Van Rooy, P., 1997. Interactieve planvorming voor waterbeheer. Proefschrift Technische Universiteit Delft, The Netherlands (in Dutch with an English summary).

Van Straten, G., 1986. Identification, uncertainty assessment and prediction in lake eutrophication. Thesis University of Twente, The Netherlands.

Van Straten, G., 1998. Models for water quality management: the problem of structural change. *Water Science and Technology* 37: 103-111.

Van Straten, G. & K.J. Keesman, 1991. Uncertainty propagation and speculation in projective forecasts of environmental change: a lake eutrophication example. *Journal of Forecasting* 10: 163-190.

Visser, P.M., B.W. Ibelings, B. Van der Meer, J. Koedood & L.R. Mur, 1996. Artificial mixing prevents nuisance blooms of the cyanobacterium *Microcystis* in Lake Nieuwe Meer, The Netherlands. *Freshwater Biology* 36: 435-450.

Vollenweider, R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD, Paris, DAS/CIS 68.27.

Vollenweider, R.A., 1969. Möglichkeiten und Grenzen elementarer Modelle der Stoffbilanz von Seen. *Archiv für Hydrobiologie* 66: 1-36.

Vollenweider, R.A., 1975. Input-output models with special reference to the phosphorus loading concept in limnology. *Schweizerische Zeitschrift für Hydrologie* 37: 53-84.

Young, P.C., 1983. The validity and credibility of models for badly defined systems. In: M.B. Beck & G. Van Straten (eds.), Uncertainty and Forecasting of Water Quality. Springer-Verlag, Berlin, pp. 69-98.

Zevenboom, W., A. Bij de Vaate & L.R. Mur, 1982. Assessment of factors limiting growth rate of Oscillatoria agardhii in hypertrophic Lake Wolderwijd, 1978, by use of physiological indicators. *Limnology and Oceanography* 27: 39-52.

Environmental model calibration under different specifications: application to the model SED

D.T. Van der Molen[1] & J. Pintér[1,2], 1993

Ecological Modelling 68: 1-19.

[1] Institute for Inland Water Management and Waste Water Treatment, Lelystad, The Netherlands
[2] Present address: Pintér Consulting Services, 129 Glenforest Dr., Halifax, N.S., Canada

Abstract

The subject of this paper is to draw attention to various criteria that can be used to calibrate a model (i.e. to assess its performance) and to study the differences in the results obtained when performing calibration using these criteria in a practical example. A recently proposed global optimisation procedure is applied for parametrising a simple dynamical model that describes the release of phosphorus from sediments of shallow, eutrophic lakes.

In general, a number of tentative discrepancy measures can be analysed, when a formal optimised calibration procedure is applied. The type of discrepancy measure depends, inter alia, on the number and type of available observations and the objective of the modelling exercise. Therefore, the results obtained should always be carefully verified. Special attention must be paid to the calibration of environmental models, as their 'soft' character frequently demands the application of non-standard discrepancy measures.

List of symbols

$t = 1,\ldots,T$	time-moments of observation
T	sample size
P_t	model output
O_t	observation (element of a homogeneous set of measured data)
D	discrepancy measure (expresses the deviation of the sequences $\{P_t\}$ and $\{O_t\}$)
X	set of feasible model parametrisations, $x \in X$ is a real n-vector
$j = 1,\ldots,J$	index of model output variables
E	expected (mean) value operator
V	variance operator
ω	weight vector (T-dimensional)
δ	tolerance level
ϕ	random variable expressing relevant uncertainties (scalar or vector, depending on the actual context)

Introduction

Environmental model development is an essential conceptual tool of the related theoretical and applied research (see, e.g., Loucks *et al.*, 1981; Haith, 1982; Novotny & Chesters, 1982; Beck & Van Straten, 1983; Jørgensen, 1983; Orlob, 1983; Beck, 1985; Somlyódy & Van Straten, 1986). Generally speaking, the following main phases of quantitative analytical environmental modelling can be distinguished:

- formulation of model objectives
- set up of model structure
- calibration
- validation
- application (analysis, forecasting, control, management).

Consequently, calibration - finding the 'best' or just 'suitable' parametrisations - is an important stage in model development.

Confronting model results with expertise, background information and available observations, the following issues have to be properly addressed:

- lack of 'perfect' scientific knowledge (and, hence, of 'perfect' models)
- complicated model structure
- necessary decomposition of reality, followed by (re)aggregation in the frame of a model
- inadequacy/errors in monitoring and data processing
- subjectivity in the interpretation of results.

According to the iterative model calibration procedure indicated by Figure 1, one can attempt to approximate the 'very best' theoretically admissible combination of the parameters which results in a model output that is 'as close as possible' to the set of available observations or, alternatively, to find 'acceptable' model parametrisations. The latter objective is especially relevant when 'soft' systems are modelled. These systems are characterised by complex and ill-defined processes, and therefore parameters have wide defensible ranges and observations cannot unequivocal be compared to the system behaviour.

In this paper particular attention will be devoted to the selection of diverse (scalar or vector, deterministic or stochastic) discrepancy measures, in order to express the 'difference' between model output and corresponding observation data. These various model forms should reflect the intended model use, as there is no 'universally best' calibration paradigm. In other words, the 'goodness' of the model is to be judged in terms of those features that are important for a particular application.

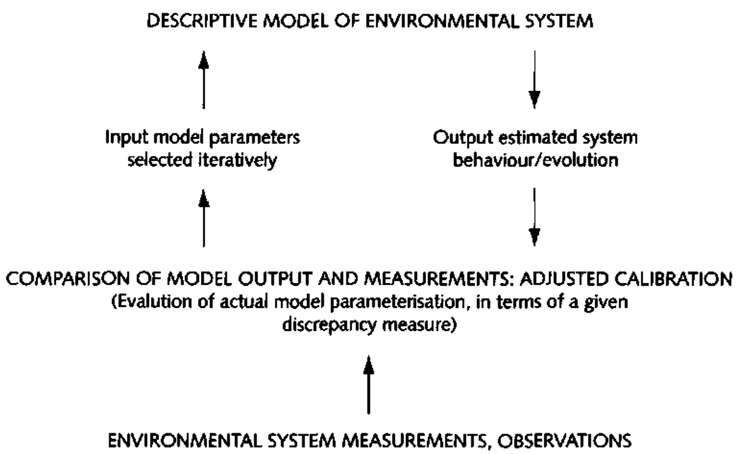


Figure 1 *Model calibration scheme (from Pintér & Van der Molen, 1991).*

Methodology and scope of application

In mathematical terms, the objective of finding the optimal model parametrisation can be expressed by the general problem statement

$$\text{minimise } D(x) \qquad D(x) := D\{P_t(x), O_t\}_{t=1,\dots,T} \qquad x \in X \tag{1}$$

Due to the frequent non-linearity of environmental models, in many cases the discrepancy measure D will be multi-extremal, with respect to the parameter vector x. In other words, initiating a 'standard' search procedure from different points of the feasible parameter domain X may often lead to markedly differing results, both in terms of the parametrisation found and its 'performance' (as being expressed by the function D). Therefore - contrary to most 'classical' model calibration approaches (see e.g. Box & Jenkins, 1970) -, such numerical methodology is to be applied that is capable to find (approximate) the 'very best' x in X. This objective necessitates the application of some properly chosen global optimisation strategy. The specific theory and numerical methodology applied here is described elsewhere (see, e.g., Pintér, 1990, 1991b, 1992).

For illustrative purposes, several calibration model variants will be formulated with respect to the model SED (Van der Molen, 1991). The relatively simple model SED involves only a few parameters to be calibrated, hence facilitating the investigation of diverse discrepancy measures.

Performance criteria

Single criterion analysis

For a single state variable, the discrepancy measure is often derived in the form of

$$D(x) := 1/T \cdot [\Sigma_{t=1,T} |P_t(x) - O_t|^{\tau}]^{1/\tau} \qquad 1 \le \tau \le \infty \tag{2}$$

The factor $1/T$ allows to compare coherently calibration results for different sample sizes $t = 1,\dots T$. The most well-known special cases of (2) include the minimisation of the (averaged) absolute deviations (3), the (averaged) least squares estimation problem (4) and the minimisation of the maximal discrepancy between the set of model output values and the observations (minimax objective function form) (5). For a related general exposition, see e.g. Braess (1986), or - in the context of environmental modelling - Thomann (1982), Pintér (1990) and references therein.

$$D(x) := 1/T \cdot \Sigma_{t=1,T} |P_t(x) - O_t| \qquad \tau = 1 \tag{3}$$

$$D(x) := 1/T \cdot [\Sigma_{t=1,T} |P_t(x) - O_t|^2]^{1/2} \qquad \tau = 2 \tag{4}$$

$$D(x) := 1/T \cdot \max_t |P_t(x) - O_t| \qquad \tau = \infty \tag{5}$$

The mathematical properties of the function (2) imply that for increasing values of τ the 'outliers' (i.e. the more significant discrepancies between model output and measurements) are more heavily 'penalised'. This way, the choices $\tau = 1$ and $\tau = \infty$ represent the two possible extreme points of view: the function form (3) includes all pairs of output and observation values with equal weight, while (5) considers only the maximal deviation. This comment may help to choose the 'right' value of the parameter τ (or better, to consider a meaningful selection of them). In order to avoid typical numerical instabilities, the use of the exact minimax objective function form (5) can be recommended (replacing the settings of 'large' values τ) in most practical cases. When applying (2) for increasing values τ, it is often necessary to smoothen the observed data to avoid dominance by (possible false) outliers.

Replacing now the absolute difference between $\{P_t\}$ and $\{O_t\}$ by its relative (to the non-zero measurements) value transforms function (2) to (6). This objective function type can be applied, when it is assumed that relative function values are preferable to their real values, as characterising the 'goodness of fit'. The underlying idea leading to this function D can be that larger absolute valued model results and measurements may also have larger inherent errors. Again, the general objective function form (6) can be specified in a number of ways, analogously to (3) - (5).

$$D(x) := 1/T \cdot [\Sigma_{t=1,T} | (P_t(x) - O_t)/O_t |^{\tau}]^{1/\tau} \qquad O_t \neq 0,\ 1 \leq \tau \leq \infty \qquad (6)$$

The class of discrepancy measures (2) is invariant with respect to translations. That is, if all model output values and observations are replaced by adding a constant to them, then none of these objective function values will change. By contrast, the function form (6) is dimensionless and invariant with respect to all multiplicative transformations, but it is not invariant with respect to translations: this way, the scale dependencies eliminated by (2) or (6) are different.

Frequently, the natural fluctuations of the investigated environmental system and/or the inherent error of the measurements are also important to consider, when evaluating the 'goodness' of diverse model parametrisations. For example, one can estimate the ratio of model discrepancy to the 'natural' (inherent measurement data) fluctuations. In such cases, the following statistically established discrepancy measure can be recommended. If $E(O_t)$ and $V(O_t)$ are the time average and the variance of the observations, respectively, then the standardised model output and observations are defined by (7) and (8) and the discrepancy function is given by (9).

$$P_t^s(x) = [P_t(x) - E(O_t)] \,/\, V(O_t)^{1/2} \qquad V(O_t) \neq 0 \qquad (7)$$

$$O_t^s = [O_t - E(O_t)] \,/\, V(O_t)^{1/2} \qquad V(O_t) \neq 0 \qquad (8)$$

$$D(x) := 1/T \cdot [\Sigma_{t=1,T} | P_t^s(x) - O_t^s |^{\tau}]^{1/\tau} \qquad 1 \leq \tau \leq \infty \qquad (9)$$

With respect to the choice of τ, analogous remarks to those on function (2) can be made. Function (9) is invariant, if linear transformations are applied (that replace the value P by $mP+c$, $m \neq 0$ and c being arbitrary real numbers). Consequently, the discrepancy measure (9) will not vary, if the scale unit and/or location parameters are changed. An example of a possible application of the function form (9) is the case when the model output and observations (of the same variable) at two different monitoring stations on a river are to be included into a single objective function. The standardising transformation (7) - (8) than 'filters out' the site-specific variances, and the objective function will be the sum of two functions of the type (9), with an appropriate scaling.

For various reasons (e.g. timing, place, reliability etc. of the measurements taken), different pairs of model output and system observation values may be considered as of different importance (although related to the same state variable). This fact can be explicitly taken into consideration by defining a coherent system of relative weights ω_t, which are incorporated into the objective function. For instance, a possible generalisation of (2) has the form (10) in which $\omega_t > 0$ and $\Sigma_{t=1,T}\, \omega_t = 1$. Function (10) reduces to (2), if $\omega_t = 1/T$ for $t = 1,...,T$. Evidently, specifications and extensions of (10) similarly to those expressed by (3) - (9) are again possible, depending on the importance of 'outliers' and the invariance structure required.

$$D(x) := [\Sigma_{t=1,T} \,|\, \omega_t \cdot (P_t(x) - O_t) \,|\, ^{\tau}]^{1/\tau} \qquad 1 \leq \tau \leq \infty \qquad (10)$$

Multiple criteria analysis

Throughout the previous section, it has been tacitly assumed that both the environmental state observations and the corresponding model output can be expressed by scalar values: hence, only a single environmental quality indicator was considered, when evaluating the discrepancy between model output and observation data. This assumption justified the formulation of the scalar optimisation problems presented. In reality, the evolution of an environmental system is often better characterised by modelling and measuring a number of environmental state descriptors. For instance, a nutrient can be modelled and observed in both the sediment and the water body of a lake system, or markedly different types of water quality indicators (nutrients, algal chlorophyll and so on) can be analysed. In such or similar cases, it is important to find proper multiple criteria calibration models. Formally stated, if $j = 1,...,J$ denote the indices of system state variables, then a suitable vector-valued discrepancy measure (11) is to be considered, in order to evaluate and compare different model parametrisations.

$$D(x) := [D^j\{P_t^{\,j}(x), O_t^{\,j}\}_{t=1,...,T} \qquad j = 1,...,J] \qquad (11)$$

The minimisation of the vector-function D in (11), generally speaking, has no well-defined sense. This fact necessitates the introduction of appropriate multiple objective optimisation type generalisations of the scalar calibration paradigm investigated above. The following brief discussion will be related to conceptually correct scalarisations for commensurable and non-commensurable variables. For additional details on multiple objective modelling and decision-making, see, e.g., the survey of Hwang & Masud (1980).

Commensurable variables are quantities that can be evaluated on the same scale. Non-commensurability implies that the system state variables of model output and measurements can not be scaled simultaneously. Observations related to the same water quality indicator, but taken in different water layers (commensurable) and different water quality indicators (non-commensurable) may serve as examples. In many cases an appropriate prior rescaling of the state variables may be necessary. For example, chloride and phosphorus measurements may be of the order 100 mg l^{-1} and 0.1 mg l^{-1} respectively: therefore a linear transformation of these values (e.g. transforming them into the dimensionless interval [0,1]) is desirable, before defining a scalarised joint discrepancy function.

Commensurability implies that weights ω^j can be introduced that express the relative importance of D^j. The most usual way of incorporating commensurable objectives into a single objective function is to define (12) (with $\omega^j > 0$ and $\Sigma_{j=1,J}\, \omega^j = 1$), which is then minimised over the feasible parameter set X to yield a 'good' model calibration option. Naturally, the solution x now depends also on the actual choice of the weight factors. While in principle any of the model specifications described in the previous section can be applied, the component objectives D^j will frequently have the same functional form.

$$D(x) := \Sigma_{j=1,J}\, \omega^j . D^j\{P_t^{\,j}(x), O_t^{\,j}\}_{t=1,\dots,T} \qquad (12)$$

A rational way of reformulating the multiple objective calibration problem in the case of non-commensurable variables is to choose 'the most important' system variable for defining the (primary) objective function. All the other variables will then be considered in explicit constraints, reflecting the corresponding componentwise calibration accuracy criteria. In mathematical terms, if $j = 1$ is chosen as the index of the principal calibration performance indicator and δ_j are given tolerance bounds, then the calibration problem is expressed by minimising in (13) under the constraints (14). In (13) and (14) it is not necessary at all to apply the same type of discrepancy measure, for all variables $j = 1,\dots,J$: consider, for example, the case when water quality indicators and sediment characteristics have to be expressed simultaneously.

$$\min D^1\{P_t^{\,1}(x), O_t^{\,1}\}_{t=1,\dots,T} \qquad (13)$$

$$D^j\{P_t^{\,j}(x), O_t^{\,j}\}_{t=1,\dots,T} \le \delta_j \qquad \text{for } j = 2,\dots,J \qquad (14)$$

Discrepancy measures for calibrating 'soft' systems

In many cases - because of our limited knowledge and/or the 'soft' character of the environmental system studied, the scarcity of observations and 'exact' information etc. - a strictly interpreted optimisation paradigm may be more of theoretical, than of practical interest. Just to mention a trivial example, it has no sense to calibrate a model that has more parameters than the number of observations. Under such circumstances, less ambitious objectives than formally 'best' model calibration may be more appropriate. The issue of 'soft modelling' versus 'formal optimisation' is extensively discussed in the literature; see e.g. Beck & Van Straten (1983), Beck (1985, 1987), Somlyódy & Van Straten (1986), Banks & Fitzpatrick (1990) or Van Straten & Keesman (1991). Let us only note here that, if there exists a possibility of quantitatively evaluating the 'goodness of fit' of an arbitrarily selected model parametrisation and there is a sufficiently detailed data/information background, then the above presented optimisation concept is justified, at least in a formal (mathematical) sense. Some constraint forms, aiming at just 'acceptable' or at 'sufficiently good' model parametrisations are (15) - (17), in which $E(O_t)$ is the (estimated) time average of the measurement data and $\delta > 0$ is some fixed tolerance parameter:

$$\max_t P_t(x) \leq \max_t O_t \qquad \min_t P_t(x) \geq \min_t O_t \tag{15}$$

$$\max_t | P_t(x)-O_t | \leq \delta \tag{16}$$

$$\max_t P_t(x) \leq E(O_t) + \delta \qquad \min_t P_t(x) \geq E(O_t) - \delta \tag{17}$$

The constraints (15) express that 'the model output trajectory be realised in the range of the (probably few) available observations': maximisation (or minimisation) on the left- and right-hand sides of the stated inequalities may be taken over sets of different cardinality. Another (in general, more strict) acceptability criterion can be to prescribe that the distance between the modelled system trajectory and the observation data be uniformly bounded (16). To provide an example, assume that a certain water quality component is modelled, but it is rarely measured. Due to spatial (horizontal and vertical) heterogeneity and observation or analytical errors, often a range δ around the measurements can be taken, instead of just considering the 'exact' data. In similar cases the model performance criterion can be based also on (17). An example of applying this feasibility requirement is the case, in which the variation of certain observations over a given time-period is (expectably) caused more by random and analytical errors, than by changes caused by processes one is interested in. This is often the case e.g. for sediment characteristics during a period shorter than a year.

A more general type of acceptance criterion in 'soft' modelling can be specified by supposing that the model results can be broadly classified as either being 'acceptable' or 'non-acceptable', without further differentiation between them (Keesman, 1989). For example, one might be especially interested in model performance, when the stated water quality standards are violated. This approach may induce not only feasibility constraints, but can also express certain partial calibration objectives. In general, 'sufficiently good' model parametrisations can be characterised by defining the set of acceptability A as

$$A = \{x \in X: D\{P_t(x), O_t\}_{t=1,\ldots,T} \leq \delta\} \qquad \delta > 0 \tag{18}$$

The specification of the discrepancy measure D defines the concrete form of (18). Suppose that T_1 is the 'the really important' subset of observations, i.e. they are to be followed closely. A suitable criterion function form can then be (18), in which the set T is to be replaced by T_1. Two special, practically important examples of defining the set T_1 are the following:

- a certain period of time is of particular interest, i.e.
  $T_1 = \{t=1,\ldots,T: t_{\min} \leq t \leq t_{\max}\}$
- it is of primary significance that the 'higher' and 'lower' measurement values are reproduced well by the model; in this case
  $T_1 = \{t=1,\ldots,T: | O_t | > \delta\} \qquad \delta > 0.$

All previously presented model performance criteria are based on a coherent timing of model output and observations (represented by the indices $t = 1,\ldots,T$). At the same time, in environmental models certain data (loads, meteorological forcing functions etc.) are often given on a daily, weekly or a decade time-scale. The larger time-steps often have a certain 'equalising' effect (possibly 'hiding away' some extremes that, on the other hand, may be followed by the model output). Similar examples raise the issue of defining some other measures of 'closeness' than those introduced earlier. For instance consider (19), based on (2), in which the objective function is defined by the distance between the measurement data and the linearly interpolated function passing through the (possibly discretised) model output points. $P_t^l(x)$ for $t = 1,...,T$ are those points of the linear interpolation curve that are 'closest' to the corresponding observation in a geometric sense. For illustration, see Figure 2 (note that the interpolation procedure suggested is scale dependent).

$$D\{P_t(x),O_t\}_{t=1,...,T} := 1/T\cdot[\Sigma_{t=1,T}|P_t^l(x)-O_t|^{\tau}]^{1/\tau} \qquad 1 \le \tau \le \infty \qquad (19)$$

Another time-related aspect of evaluating dynamic environmental models is that, the time-step in the calculations can be an hour, a day, a week, a month, a year or even more. For example, wind conditions can be observed hourly, while sediment characteristics are often described on a yearly time-scale. Therefore, when used simultaneously, some of the available data have to be aggregated and, evidently, the length of the aggregated time-step limits the precision of describing the temporal evolution. This comment implies that apart from variations and discrepancies of the model output and measurements the role of the time-step chosen should also be analysed when its relevance is felt. A straightforward way of doing that is to compare the 'optimal' parametrisations obtained, when applying a sequence of 'appropriate' time-steps in the model.

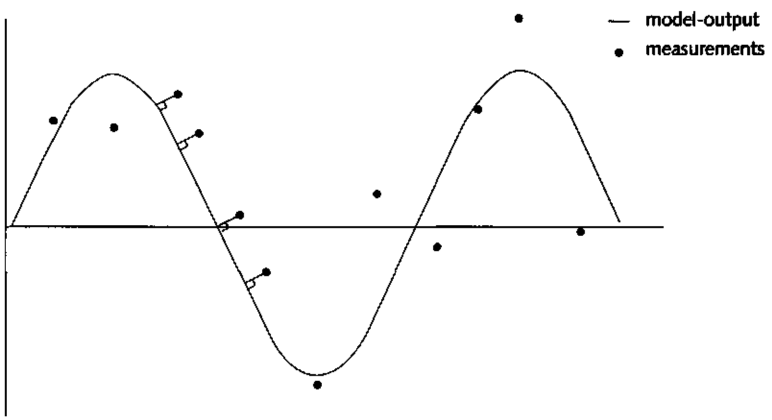


Figure 2 *Linearly interpolated model output and the points associated with the corresponding observations.*

Closing this section, we emphasise that the acceptability criterion functions presented above do not necessarily require per se the application of any formal optimisation approach. Notwithstanding, to find just admissible parametrisations can be a non-trivial issue and the (global or more conventional) optimisation approaches referred to earlier can be applied to finding feasible parametrisations.

Stochasticity aspects

The discrepancy function forms analysed above illustrate a fairly broad choice of possible basic deterministic calibration model types. At the same time, the proper consideration and handling of system-inherent uncertainties is an often neglected but essential stage in environmental modelling and management (see e.g. Beck & Van Straten, 1983; Beck, 1987; Pintér, 1991b). Assume that the uncertainties and statistical fluctuations of the studied environmental system can be modelled in a suitable probabilistic frame. For illustrating this point see Figure 3 in which the 'noisy' measurement data are modelled as mean values of normally distributed random variables. Without going into much details, we will highlight several stochastic optimisation model forms which can serve as correct generalisations of the basic calibration model (1).

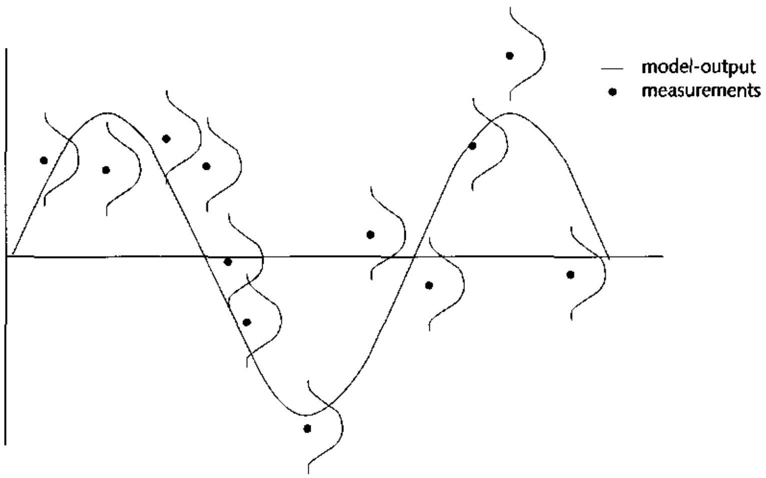


Figure 3 *Observations modelled as mean values of normally distributed random variables.*

Random (uncertain, unforeseen) events can be introduced in the discrepancy measure $D(x)$ by the realisations of a corresponding (scalar or vector) random variable ϕ:

$$\text{minimise } D(x,\phi) \qquad D(x,\phi) := D\{P_t(x,\phi), O_t(\phi)\} \qquad x \in X \tag{20}$$

This problem statement does not have a well-defined mathematical sense, since $D(x,\phi)$ is also a random variable. Notwithstanding, a large number of stochastic model variants can be interpreted as appropriate extensions of (1). A frequently used variant is the expected value model of discrepancy $E\{D(x,\phi)\}$ in which E denotes the expected (mean) value operator. In words: 'the average value of the discrepancy measure D is to be minimised'. The use of this criterion can be recommended e.g. in the case when, due to uncertainty about some model factors and parameters (that are not subject to calibration), the simulated model output exhibits stochastic fluctuations. To provide another example, the observations may be subject to random errors that have a known (modelled) statistical description (recall Figure 3). In such or similar cases, instead of finding the 'best' model parametrisation with respect to a particular realisation of the random factors involved, one may prefer the calibration which performs 'best on the average'. An extension of this model variant is the expected value-variance model of discrepancy (21).

$$\alpha \cdot E\{D(x,\phi)\} + (1-\alpha) \cdot V^{1/2}\{D(x,\phi)\} \qquad 0 \le \alpha \le 1 \text{ fixed} \tag{21}$$

$V\{D(x, \phi)\}$ denotes the variance of the random variable $\{D(x, \phi)\}$. For motivating (21), let us remark that in many cases the value of the random variable $\{D(x,\phi)\}$ may vary significantly. Therefore it may be desirable to consider not only the expected performance of an arbitrary model parametrisation but also the fluctuations of that performance. The more stability of performance is required, the smaller value of α is to be chosen. Setting $\alpha = 0$ corresponds to considering only the standard deviation of the stochastic calibration objective function, while $\alpha = 1$ deduces (21) to the expected value function. Therefore it may be informative to vary α between 0 and 1 and observe the resulting model parametrisations.

As for another example, a probabilistic target level of discrepancy can be defined (22). Let Prob denote the probability of the imbedded random event and D_{max} the (chosen) admissible maximal discrepancy between model output and observations. This way, the probability of arriving at 'acceptable' model calibration is being maximised. Again, it is reasonable to vary systematically the model parameter value D_{max} and trace (typically by Monte Carlo simulation) the corresponding changes in the calibration results.

$$\max \text{Prob}\,\{D(x, \phi) \le D_{max}\} \tag{22}$$

The numerical solution methodology chosen might vary considerably, depending on the stochastic model form(s) applied. The most typical solution methods are the following:

- diverse (analytical or discretisation-based) deterministic approximations
- selective (parametric) sensitivity analysis
- stochastic (Monte Carlo) simulation
- sequential combination of optimisation and simulation/approximation steps.

For additional details on stochastic modelling and optimisation techniques, see e.g., the surveys of Wets (1983), Beck (1987) or Pintér (1991a).

Calibration of the model SED: some illustrative results

SED is a simple, dynamical model that was recently developed for describing the release of phosphorus from sediments in shallow, eutrophic lakes (Van der Molen, 1991). The slow response of the release of phosphorus from the sediments to the overlying water, after a decrease of the external loading, is generally seen as the explanation of the disappointing results of lake recovery. The model SED includes only two state variables: dissolved and organic particulate sediment phosphorus. The latter fraction is added to the sediment by sedimentation from the overlying water and is removed by mineralisation. Mineralisation of organic phosphorus is the main source of dissolved phosphorus. Further, dissolved phosphorus is subject to ad- and desorption and can be released to the overlying water. The differential equations are solved numerically.

The model SED is calibrated on the basis of data for Lake Veluwe, The Netherlands, in which detailed monthly mass balances have been determined over the years 1978 - 1983. Calibration of the modelled phosphorus release process alone is not possible, because direct measurements of the release are rare. The difference between sedimentation (model input, based on measured chlorophyll concentration) and the release of phosphorus from the sediments is the retention of phosphorus. The calculated retention is compared with the residual term of the in-lake phosphorus mass balance. The residual term named is the difference between input, output, storage and errors in the mass balance terms: hence, it is a 'lumped' measure for the aggregation of all in-lake processes considered. In certain months, errors in the storage term are responsible for an extremely high residual, followed by an extremely low residual term in the next month (or conversely). To reduce this effect, all residual terms are averaged together with the preceding and next values, thus taking the moving average of three subsequent data. The diverse calibration results are compared with the nominal results, i.e. results obtained previously by 'visual judgement'. Only some selected calibration results are presented here, more details are reported in Pintér & Van der Molen (1991).

In Figure 4 the 'optimal' model calibration results are shown when objective functions (3) and (5) were applied. The discrepancy measure for absolute values was preferred over the use of relative values, because the extremes are of special interest from the management point of view. For the same reason, the results for several values of τ were analysed. As can be seen, some of the results obtained led to a good match between the model output and the set of observations, but some other state variables - which are not represented explicitly by the objective function form applied here - took unrealistic values. This indicated that 'wrong' model processes were dominating the calibration. This fact calls for some sort of multiple objective model formulation. Therefore, constraints were included with respect to the additional state variables: violating any of these constraints resulted in the rejection of the parametrisation result. Partly because of the above findings with respect to the relative error objective function forms, in the present numerical exercise we did not accomplish calculations using the standardised error (9) as a discrepancy measure.

In the next stage, calculations were carried out that placed a primary emphasis on the peak measurements. For example, management might be most interested in situations when maintaining

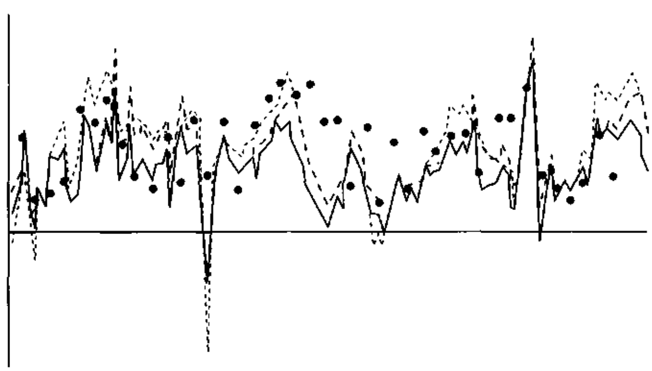


Figure 4 *Observations (dots) and calibration results: nominal values (dashed line) and application of the criteria (3) (small lines) and (5) (continuous line).*

certain environmental quality standards is the primary issue. Specifically, recalling the related discussion, the corresponding subset T_1 of the observation time-moments T was considered and $\delta = 2$ was chosen as the accuracy threshold value. Further, the objective function was based on the maximum error criterion (5). As it was expected, the model fit with respect to the 'peaks' became appreciably better, see Figure 5.

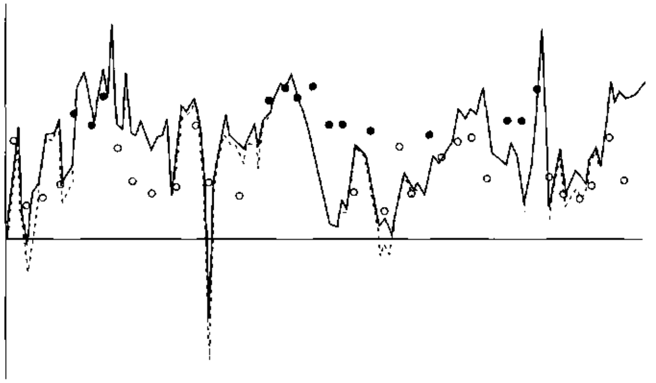


Figure 5 *Calibration with respect to a restricted set of observations: nominal (dashed line) and optimised (continuous line) model output, observations considered (full dots) and omitted (empty dots).*

In the following set of experimental runs, we assumed that the larger absolute value measurements may also be subject to larger stochastic errors. Specifically, we supposed that all measurement values follow normal distribution with expected mean $E(O_t) = O_t$ and standard deviation $V^{1/2}(O_t) = \beta \cdot O_t$, in which β is a 'noise' parameter ($\beta > 0$). From (3) and (20) the generic objective function form (23) can be derived. In this case ϕ can be seen as the (normal) measurement data error.

$$D(x,\phi) := 1/T \cdot \Sigma_{t=1,T} \mid P_t(x) - O_t(\phi) \mid \tag{23}$$

Hence, $D(x,\phi)$ will have statistical fluctuations. To illustrate this point, the empirical order statistics of the stochastic discrepancy function values, based on 200 Monte Carlo simulation cycles, is shown for a fixed set of model parameters x and different values of β (Figure 6). As it can be deduced from the empirical distributions derived (graphically or applying statistical criteria), the discrepancy function values can be assumed to (approximately) follow normal distributions. Both the mean and the standard deviation of the random discrepancy function value are increasing functions of the noise parameter β. Even a 'moderate' noise may cause large fluctuations in the performance of the model parametrisation chosen. We will not investigate here the changes in the 'optimal' parametrisation, as a function of the noise structure.

In order to obtain some preliminary insight into the expected characteristics of the performance criterion (21), for illustration three sequences of calculations were performed. In these the model parameter vector x was tentatively chosen as 'lower', 'middle' and 'upper'. For each of these vectors

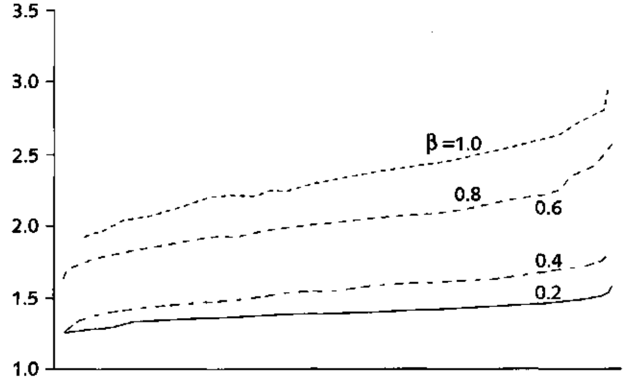


Figure 6 *Statistical variation of the discrepancy function for different values of the noise parameter β(based on 200 Monte Carlo simulation cycles).*

and for different values of β, 100 Monte Carlo cycles were accomplished, leading to corresponding sets of $D(x,\phi)$, their estimated mean and standard deviation values. These results are summarised in Figure 7. As it can be seen, the figure support the following conclusions:

- the expected value of the discrepancy function can be modelled by a power function of the (actual) noise structure,
- the standard deviation of the discrepancy function can be modelled by a linear function of the (actual) noise structure,
- the increasing observation noise, modelled by larger values of β, tends to diminish the difference between the diverse ('good' or 'bad') parameter combinations, as the better results are gradually more distorted. In contrast to the first two conclusions, this finding is more generally valid.

According to these numerically derived conclusions, the dependence of the function constants $c1$ - $c3$ in (24) on the model parameters x does not seem to be very strong. Therefore, in this case we can approximate (21) as

$$\alpha \cdot E\{D(x,\phi)\} + (1-\alpha)\cdot V^{1/2}\{D(x,\phi)\}$$

$$\approx \alpha \cdot D(x,0)\cdot[1+c1(x)\cdot\beta^{c2(x)}] + (1-\alpha)\cdot c3(x)\cdot\beta$$

$$\approx \alpha \cdot D(x,0)\cdot[1+c1\cdot\beta^{c2}] + (1-\alpha)\cdot c3\cdot\beta \qquad (24)$$

The postulated weak dependence on x implies that in this particular case the stochastic optimisation model form (21) can be approximated by minimising $D(x) = D(x,0)$ over the feasible set of model parametrisations. Of course, the weak dependence assumption should be checked thoroughly, but if it holds, the simpler deterministic model form may prove to be a good approximation to the stochastic criterion (21).

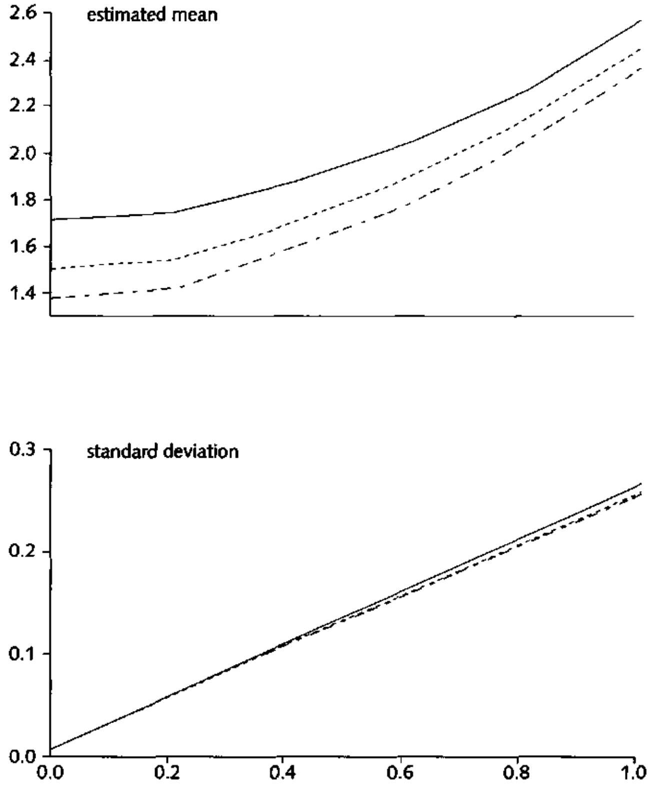


Figure 7 *Variation of the estimated mean value (upper part) and the standard deviation (lower part) of the discrepancy function for three different parameter sets, under different noise conditions (based on 100 Monte Carlo cycles in each case).*

Conclusions

As it was demonstrated, in general, a number of tentative discrepancy measures are to be analysed, when an optimised calibration procedure is applied. Further on, the results obtained should be carefully checked. A special attention must be paid to environmental models, as their 'soft' character frequently necessitates the application of non-standard discrepancy measures and corresponding calibration problem statements. To find suitable discrepancy measures one has to answer several questions, e.g.:

- which variables are most important for the model application?
- is it sufficient to know the range in which a variable varies?
- what is the importance of the variations in time of a variable?
- what to do with outliers in the observations?

This will often result in an 'acceptable' model parametrisation under the conditions specified, rather than the 'very best' theoretically admissible combination of model parameters.

The practical significance of the model SED and the available expert knowledge made possible the fast comparative evaluation of a number of the calibration model variants studied. This way, quantitative information could be gained with respect to the choice of preferable model variants that lead to realistic simulated output. On average, the 'best' calibration results improved by 10 - 20% compared with the results obtained by 'visually checked' calibration. As a result of the calibration procedure, some previously applied model parameter values changed significantly, although they still remained within their range known from laboratory experiments and literature.

In case of SED, a systematic evaluation of diverse scalar deterministic, multiple criteria and stochastic discrepancy measure forms implied a non-trivial, but still manageable computational effort. Generally speaking, the proper consideration of model stochasticities leads to a significant increase of the computational demands. In return, the 'optimal' calibration results obtained are typically more stable (robust), realistic and informative, than their simpler deterministic model based counterparts. By carrying out an appropriate off-line statistical analysis, one may be able to solve approximately the stochastic calibration problem, without accomplishing a (computationally very demanding) iterative optimisation-simulation procedure.

Acknowledgements

The research summarised in this paper was supported by the Institute for Inland Water Management and Waste Water Treatment (RIZA), Lelystad, The Netherlands. The authors wish to thank Prof. Gerrit van Straten for many constructive comments on the draft version of this paper.

References

Banks, H.T. & B.G. Fitzpatrick, 1990. Statistical methods for model comparison in parameter estimation problems for distributed systems. *Journal of Mathematical Biologie* 28: 501-527.

Beck, M.B., 1985. Water Quality Management: A Review of the Development and Application of Mathematical Models. Springer-Verlag, Berlin.

Beck, M.B., 1987. Water quality modelling: A review of the analysis of uncertainty. *Water Resources Research* 23: 1393-1442.

Beck, M.B. & G. Van Straten (eds.), 1983. Uncertainty and Forecasting in Water Quality. Springer-Verlag, Berlin.

Box, G.E.P. & G.M. Jenkins, 1970. Time Series Analysis: Forecasting and Control. Holden Day, San Francisco, CA.

Braess, D., 1986. Nonlinear Approximation Theory. Springer-Verlag, Berlin.

Haith, D.A., 1982. Environmental Systems Optimization. Wiley, New York, NY.

Hwang, C.L. & A.S.M. Masud, 1980. Multiple Objective Decision Making Methods and Applications. Springer-Verlag, Berlin.

Jørgensen, S.E. (ed.), 1983. Application of Ecological Modelling in Environmental Management. Elsevier Scientific Publishing Company, Amsterdam.

Keesman, K.J., 1989. A set-membership approach to the identification and prediction of ill-defined systems: application to a water quality system. Thesis University of Twente, Netherlands.

Loucks, D.P., J.R. Stedinger & D.A. Haith, 1981. Water Resources Systems Planning and Analysis. Prentice-Hall, Englewood Cliffs, NJ.

Novotny, W. & G. Chesters, 1982. Handbook of Nonpoint Pollution. Van Nostrand Reinhold, New York, NY.

Orlob, G.T. (ed.), 1983. Mathematical Modeling of Water Quality: Streams, Lakes and Reservoirs. IIASA-Wiley, New York, NY.

Pintér, J., 1990. Globally optimized calibration of environmental models. *Annals of Operations Research* 25: 211-222.

Pintér, J., 1991a. Stochastic modelling and optimization for environmental management. *Annals of Operations Research* 31: 527-544.

Pintér, J., 1991b. Lipschitzian global optimization: Some prospective applications. In: C.A. Floudas & P.M. Pardalos (eds.), Recent Advances in Global Optimization. Princeton University Press, Princeton, NJ, pp. 399-432.

Pintér, J., 1992. Convergence qualification of adaptive partition algorithms in global optimization. *Mathematical Programming* 56: 343-360.

Pintér, J. & D.T. Van der Molen, 1991. Calibration under different problem specifications: Application to the model SED. RIZA nota 91.049, Lelystad.

Somlyódy, L. & G. Van Straten (eds.), 1986. Modeling and Managing Shallow Lake Eutrophication. Springer-Verlag, Berlin.

Thomann, R.V., 1982. Verification of water quality models. *Journal of Environmental Engineering* 108 (EE5): 923-940.

Van der Molen, D.T., 1991. A simple, dynamic model for the simulation of the release of phosphorus from sediments in shallow, eutrophic systems. *Water Research* 25: 737-744.

Van Straten, G. & K.J. Keesman, 1991. Uncertainty propagation and speculation in projective forecasts of environmental change - a lake eutrophication example. *Journal of Forecasting* 10: 163-190.

Wets, R., 1983. Stochastic programming: solution techniques and approximation schemes. In: A. Bachem, M. Grötschel & B. Korte (eds.), Mathematical Programming: The State of the Art. Springer-Verlag, Berlin, pp. 556-603.

Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading

D. T. Van der Molen[1] & P.C.M. Boers[1], 1994

In: E. Mortensen, E. Jeppesen, M. Søndergaard & L.K. Nielsen (eds.), Nutrient Dynamics and Biological Structure in Shallow Freshwater and Brackish Lakes. Proceedings International Conference, 16-20 August 1992, Silkeborg, Denmark. *Hydrobiologia* 275/276: 379-389.

1 Institute for Inland Water Management and Waste Water Treatment, Lelystad, The Netherlands

Abstract

An analysis of data from 49 shallow lakes showed, that the parameters of empirical models between phosphorus loading and concentration in the lake (e.g. Vollenweider type of relations) differ significantly for lakes without or with a reduced external loading. For lakes without a reduction of the external loading the summer phosphorus concentration is determined by the external phosphorus loading and the hydraulic loading. For these lakes the 'classical' models suffice; deviations between calculations and measurements are partly due to errors made in the determination of the loading.

In contrast, for lakes where the external loading was reduced, the measured internal loading explains most of the variation in the summer lake concentration. The external loading is of minor importance and the 'classical' models cannot be applied. The internal loading measured before reduction of the external loading is not useful in predicting the concentration afterwards. Instead of the internal loading, the sediment composition can be used. The advantage of using sediment composition is that these variables are easier to determine and vary less in time. The most promising variable is the ratio between total P and total Fe in the sediment.

Abbreviations

Q_s	hydraulic loading (m y^{-1})
σ	hydraulic retention time (y)
L_{ext}	external phosphorus loading (gP m^{-2} y^{-1})
L_{int}	internal phosphorus loading (gP m^{-2} y^{-1})
P_{lake}	phosphorus concentration in the lake (gP m^{-3})
P_{inlet}	phosphorus concentration in the inlet water (gP m^{-3})
P_{sed}	phosphorus content of the sediment (gP kg^{-1} dry weight)
Fe_{sed}	iron content of the sediment (gFe kg^{-1} dry weight)
Y	dependent variable multiple regression calculations
$X1$, $X2$	independent variables multiple regression calculations
a, $a1$, $a2$, b	constants

Introduction

Eutrophication is still a major water quality problem. Although harmful effects of eutrophication extend from freshwater lakes to brackish waters and parts of the sea, in the past decade an increasing amount of documentation has appeared on recovering lakes (see for example Moss *et al.*, 1986; Hosper & Meijer, 1986; Cooke *et al.*, 1986; Hosper, 1989; Sas, 1989; Jeppesen *et al.*, 1991; Jagtman *et al.*, 1992). Several measures are being taken, but the most important is still the reduction of nutrient loading. Phosphorus (P) is regarded as the main factor controlling eutrophication in lakes.

Empirical relations between the external loading of P and water and the P concentration in the lake are widely used (Vollenweider, 1975; Vollenweider & Kerekes, 1982). The relations are based on several assumptions, e.g., a well mixed lake and equilibrium between water and sediments. The relations are affected by broad confidence limits and limited applicability (Reckhow & Chapra, 1979). For example, the model cannot be applied to the transient phase of a lake that occurs after the start of a restoration program, because the internal loading is not in equilibrium with the external loading (Lijklema, 1986; Sas, 1989). Data on recovering lakes are only recently available, but the 'transient phase' is still hard to define. Jeppesen *et al.* (1991) showed a broad range in the duration of the recovery phase and discussed factors which are expected to account for the delay in recovery.

To increase the applicability of empirical models between the external loading of P and water and the P concentration in lakes, we included the measured internal loading or the sediment composition as independent variables in the data analysis. Only shallow lakes were examined. A difference was made between lakes in which the external P loading is decreased due to measures and lakes without these measures.

Material and methods

General description of the data

Data on 49 shallow (mean depth less than 6 m) lakes were gathered by literature survey and additional measurements (see Appendix). In many cases the literature was 'grey', so no references could be made. Additional information about the origin of the data is published elsewhere (Boers & Van der Molen, 1992). When data were available before and after a reduced external P loading, lakes appear twice in the data set. Summer (April - September) averaged values were used for external and internal P loading, hydraulic retention time and P concentration in the water, because this growing season is the most important period of the year. If available and if relevant, the data were averaged for the lake using surface area weight factors. The data of most variables are not normally distributed, but the logarithms of the data show more or less a normal distribution (Figure 1).

Data on loading and sediment composition

Data on external loading vary from detailed mass balances, e.g. Veluwemeer, to global estimates, as in case of Lillesjön. Especially data on the loading of water and P are hampered with errors. The P concentration in the water is a relatively reliable parameter, although it is possible that incidentally an unrepresentative part of the lake has been sampled.

The P release from sediments can be determined in several ways (Holdren & Armstrong, 1980; Boström, 1984; Boers *et al.*, 1984). The additional measurements in this study were carried out with a continuous flow system according to Boers & Van Hese (1988). In this procedure the release rates are averaged over the first 28 days, in which easily degradable organic phosphorus may be exhausted. However, Boers & Van Hese (1988) did not measure a significant decrease in release during 28 days and Jensen *et al.* (1992) found a good correlation between initial release rates (averaged over 2 days) and the averaged release over a prolonged period (21 days). Literature data concerning batch experiments were, if necessary, recalculated to initial release rates, because a build-up of phosphate in the overlying water decreases the release of P after some time (Boers & Van Hese, 1988). Apart from different methods, the conditions can also vary significantly. As far as possible we used data measured at 20 °C, without pH manipulation and under oxidised conditions.

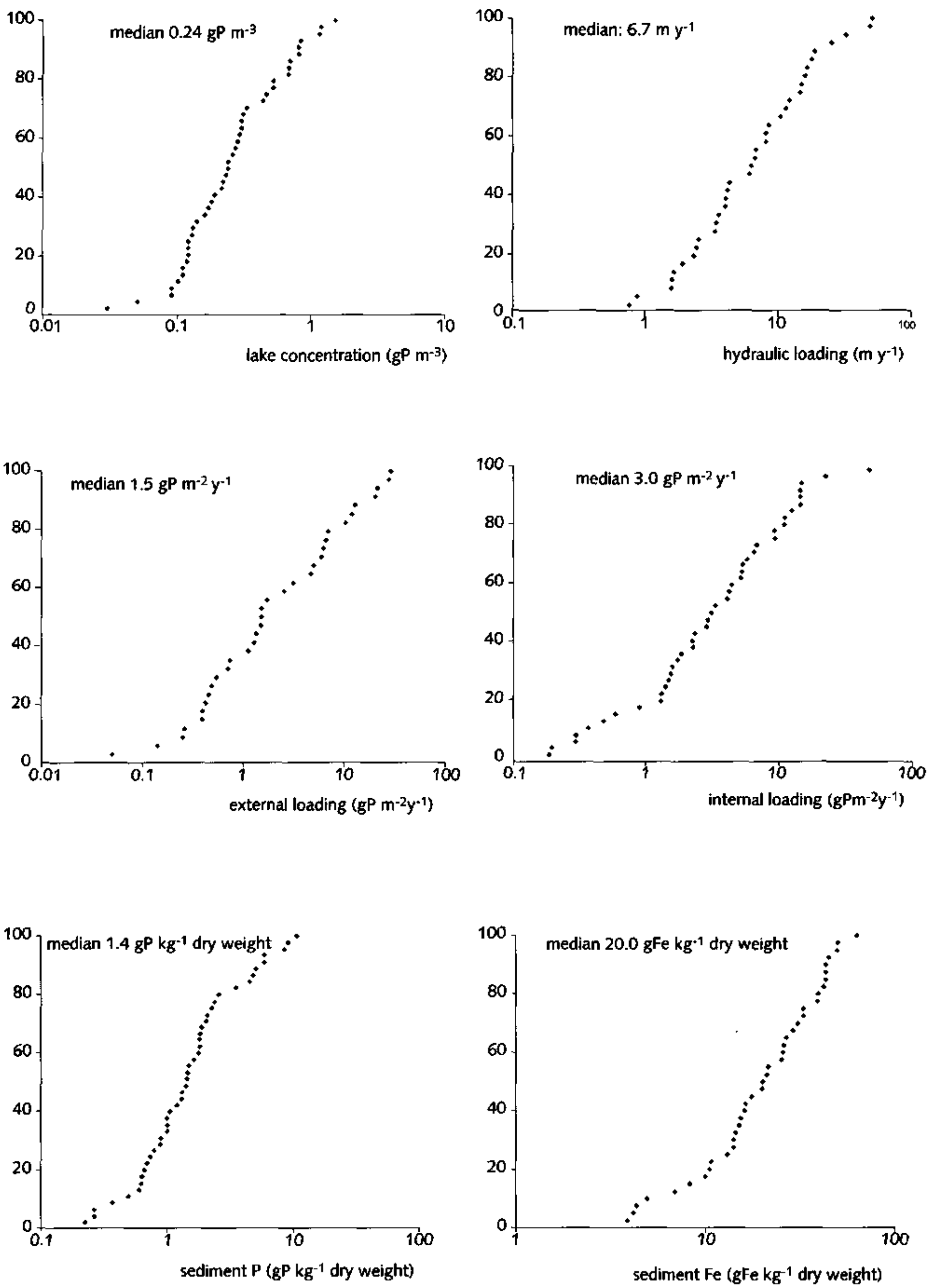


Figure 1 *Frequency distributions of the main variables of the data analysis.*

Several authors looked for relations between P (-fractions) in the sediment and the release of P from sediments (Boström, 1984; Nürnberg, 1988; Shaw & Prepas, 1990; Jensen *et al.*, 1992). Jensen *et al.* (1992) also found a direct relation between the sediment composition and the concentration in the water. Because Fe is generally seen as the dominant component for the adsorption capacity of P (Williams *et al.*, 1971; Emerson, 1976), both the total P content and the ratio between the P and Fe content in the sediment were used to characterise the sediment. Different methods to determine Fe and P were not distinguished. If available, sediment inorganic P fractions, according to the extraction scheme of Hieltjes & Lijklema (1980), were gathered (not shown in the Appendix). The extraction scheme discriminates between loosely bound P, iron and aluminum bound P and P incorporated in calcium minerals.

Calculation methods

Two approaches were followed in order to obtain simple relations between P loading and concentration. The first resembles the well-known Vollenweider approach. Vollenweider (1975, 1979) and Vollenweider & Kerekes (1982) derived relations between the P concentration in the lake and the P concentration of the inlet water, corrected for the hydraulic retention time (1). The shape of this equation is derived from a steady state mass balance and a first order removal of phosphorus in the water column (Larsen & Mercier, 1976). To decrease the effect of extreme values, all calculations are performed with the logarithms of the data. After log transformation, equation (1) becomes (2).

$$P_{lake} = b \cdot (P_{inlet} / (1+\sqrt{\sigma}))^{a} \qquad (1)$$

$$\log P_{lake} = a \cdot \log (P_{inlet} / (1+\sqrt{\sigma})) + \log b \qquad (2)$$

P_{inlet} equals external P loading divided by hydraulic loading. Internal loading was introduced in a similar way by dividing L_{int} by Q_s.

Multiple regression was used to find out if addition of a variable improves the relation or if a variable can be replaced by another variable. Again, the calculations were performed with the logarithms of the data (3). A disadvantage of the use of multiple regression is that the equation is not derived from a mass balance. Suppose Y is the concentration in a lake and the independent variables $X1$ and $X2$ are two mass fluxes. The resulting product of two fluxes has no meaning (4).

$$\log Y = a1 \cdot \log X1 + a2 \cdot \log X2 + ... + b \qquad (3)$$

$$Y = 10^{b} \cdot X1^{a1} \cdot X2^{a2} \qquad (4)$$

Results

Relations between external or internal P loading and P concentration in the lake

In the Vollenweider type of relations only external loading is taken into account by P_{inlet}. We applied equation (2) for external loading, internal loading and the sum of both in order to estimate the P concentration in the water and to compare this with measured concentrations. The incorporation of L_{int} is described in the previous paragraph.

It appears that the P concentration in lakes without reduction of the external loading is almost completely determined by the external loading, corrected for the hydraulic loading (Table 1).

Table 1 *Correlation (r^2) between the measured lake P concentration and the 'inlet' P, corrected for the hydraulic retention time according to equation (2). ** $p<0.01$*

P_{inlet}	L_{ext} / Q_s	L_{int} / Q_s	$(L_{ext}+L_{int}) / Q_s$
All lakes (n=33)	0.36^{**}	0.49^{**}	0.54^{**}
Lakes without reduction (n=23)	0.88^{**}	0.45^{**}	0.79^{**}
Lakes with reduction (n=10)	0.16	0.88^{**}	0.84^{**}

Furthermore, the external and the internal P loading correlate only for these lakes ($\log L_{int} = 0.85 \cdot \log L_{ext} - 0.04$; n=23, r^2=0.58, $p<0.01$) (Figure 2). After reduction of the external loading, there is a good correlation between the measured internal loading and the P concentration in the lake. When the Vollenweider type of relation is applied to 'all lakes', internal loading gives a better correlation than external loading, while the sum of external and internal gives the highest correlation. The results of using the sum of L_{ext} and L_{int} improved, when both variables were corrected for hydraulic retention time, instead of only the sum of the loadings (not shown in Table 1).

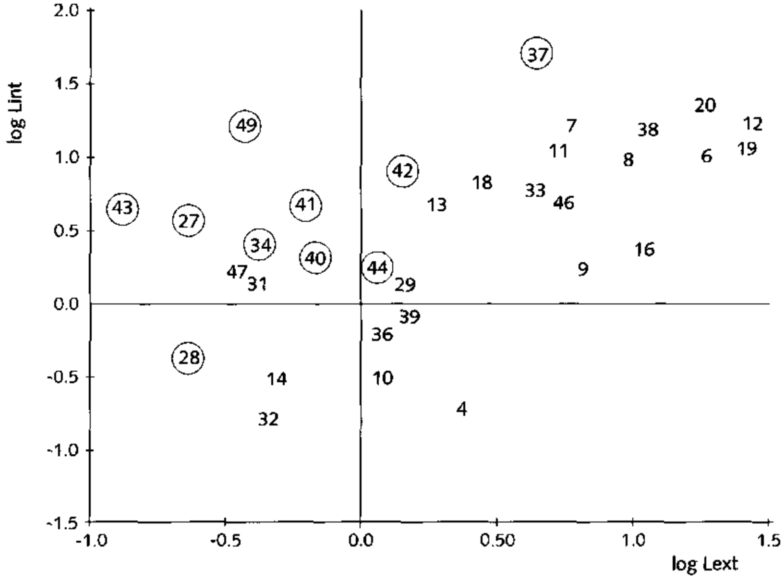


Figure 2 *Relation between the logarithms of the external and internal P loading (numbers refer to the Appendix, circles indicate reduced external loading).*

Similar results are derived by using multiple regression. Combinations of external loading, internal loading and hydraulic loading are used as independent variables to estimate the P concentration in the lake (5).

$$\log P_{lake} = a1 \cdot \log L_{ext} + a2 \cdot \log L_{int} + a3 \cdot \log Q_s + b \quad (5)$$

The external loading explains most of the variation in the P concentration for lakes without reduced external loading, while the internal loading is the most important factor in explaining the variation in the P concentration for lakes with a reduced external loading (Table 2).

Table 2 *Variation in the measured P concentration explained by the external loading (a1), the internal loading (a2) and the hydraulic loading (a3), according to equation (5).* ** *p<0.01*

	r^2	*a1*	*a2*	*a3*	*b*
Lakes without	0.62**	0.50			-0.87
reduction P	0.89**	1.02		-0.85	-0.38
loading (*n*=23)	0.91**	0.88	0.14	-0.81	-0.40
Lakes with	0.06	0.17			-0.55
reduction P	0.17	0.38		-0.37	-0.25
loading (*n*=10)	0.85**	0.04	0.57	-0.41	-0.

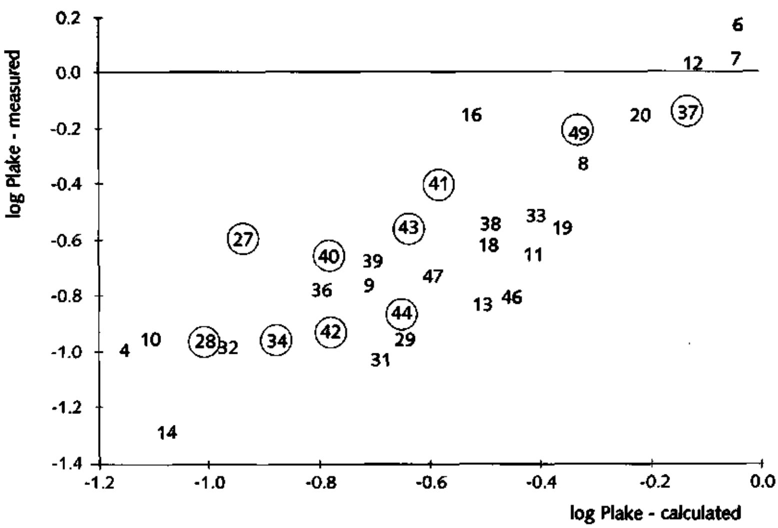


Figure 3 *The logarithm of the measured P concentration against the logarithm of the P concentration calculated by multiple regression with external P loading, internal P loading and hydraulic loading as independent variables (numbers refer to the Appendix, circles indicate reduced external loading).*

The hydraulic loading is more important for lakes without reduction than for lakes with reduction of the external loading. The absolute value of the coefficient for hydraulic loading (*a*3) is almost equal to the dominating coefficient for P loading (*a*1 or *a*2). This points to a relation between the concentration in the lake and the concentration in the inlet water (L_{ext}/Q_s or L_{int}/Q_s), rather than to the loading. For lakes without a reduction of the external loading there is almost a log-linear relation between the lake P concentration and L_{ext} or L_{ext}/Q_s. After reduction of the external loading the response of the lake concentration to loading is weaker. Figure 3 shows the results of the measured P concentration and the P concentration calculated with data from external, internal and hydraulic loading for all lakes.

64

Relations between sediment composition and P concentration in the lake or internal P loading

Table 3 presents the results of the simple regression analysis between the sediment composition and the P concentration in the water or the measured internal loading as dependent variable. The sediment composition is described by the P content and the ratio between P and Fe. Although the correlations are weak, some conclusions can be drawn:

- The relations are always better for the P concentration in the lake than for the measured internal loading.
- The available P, expressed as the P/Fe ratio, results in better correlations than the P content of the sediment.
- In the relation between the P/Fe ratio and the lake P concentration two groups of P/Fe ratios can be distinguished; the division is around 0.08 gP/gFe (Figure 4). The relatively good correlation between the two variables is caused by the higher values of this P/Fe ratio.
- In the relation between the P/Fe ratio and the internal loading no significant difference was found by dividing the data of the P/Fe ratio in two groups (not shown in Table 3).
- In all cases the correlations were better for lakes after reduction than for lakes that have not been treated (only shown for the relation of P_{sed}/Fe_{sed} and P_{lake} in Table 3). The differences were larger for relations with L_{int} than with P_{lake}.

Table 3 *Correlation between the measured internal P loading or the lake P concentration (Y) and the P content or the P/Fe ratio in the sediment (X), according to equation (3).* ** *p<0.01,* * *0.01<p<0.05*

Y	*X*	*n*	r^2	*a*	*b*
L_{int}	P_{sed}	43	0.25**	0.79	0.29
L_{int}	P_{sed}/Fe_{sed}	38	0.29**	1.29	1.86
P_{lake}	P_{sed}	38	0.29**	0.48	-0.68
P_{lake}	P_{sed}/Fe_{sed}	33	0.53**	0.99	0.49
P_{lake}	P_{sed}/Fe_{sed} (<0.08 gP/gFe)	21	0.09	0.57	-0.09
P_{lake}	P_{sed}/Fe_{sed} (>0.08 gP/gFe)	12	0.53**	1.39	0.84
P_{lake}	P_{sed}/Fe_{sed} (without reduction)	20	0.53**	0.90	0.38
P_{lake}	P_{sed}/Fe_{sed} (with reduction)	8	0.62*	0.97	0.51

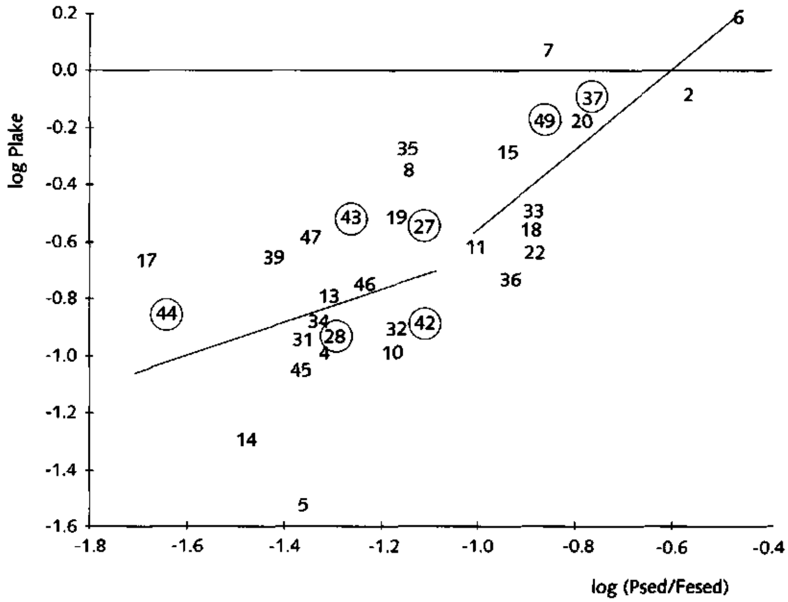


Figure 4 *The P/Fe ratio against the P concentration in the lake (numbers refer to the Appendix, circles indicate reduced external loading and the two lines represent the regression equations for P_{sed}/Fe_{sed} < and > than 0.08 gP/gFe).*

The total P content of the sediment correlated well with the sum of the fractions liberated by the extraction according to Hieltjes & Lijklema (r^2=0.69, n=30, p<0.01). On average, the sum of the fractions makes up 50% of the total P. Combinations of fractions of the extraction scheme correlated better with the internal loading, but worse with the concentration in the lake compared to the P content of the sediment or the P/Fe ratio. Of all possible combinations the sum of loosely bound and iron and aluminum bound P resulted in the best correlations (Table 4).

Table 4 *Correlation between the measured internal P loading or the lake P concentration (Y) and the sum of loosely bound and iron and aluminum bound P in the sediment (X), according to equation (3).* ** *p<0.01*

Y	*X*	*n*	r^2	*a*	*b*
L_{int}	Loosely- and Fe, Al bound P	30	0.58**	1.00	0.67
P_{lake}	Loosely- and Fe, Al bound P	26	0.46**	0.50	-0.50

Integration of sediment composition in the relations between external or internal P loading and the P concentration in the lake

Previous calculations showed that the use of internal P loading in addition to or instead of external loading, improves the predictive power of the relation between loading and concentration, in particular for lakes with a reduction of the external loading. Furthermore, the sediment composition correlated significantly with the P concentration in the lake. The P content of the sediment is determined by several factors, e.g., external P loading, hydraulic loading and the duration of the loading. Also dilution with other material may be important, especially when P adsorbing compounds are involved. The sediment composition has a minor annual variation and changes only slowly on longer time scales compared to the internal loading (Lijklema, 1986). Therefore multiple regression calculations were performed with sediment composition (P content and the ratio P/Fe) instead of internal loading to improve the relation between P loading and concentration in the lake (Table 5).

Table 5 *Correlation (r^2) between the measured lake P concentration (Y) and the external P loading (X1), hydraulic loading (X2) and internal loading or a measure for the internal loading (X3), according to equation (3).* ** *$p<0.01$,* * *$0.01<p<0.05$*

X3 =	L_{int}	P_{sed}	P_{sed}/Fe_{sed}	None
All lakes (*n*=28)	0.77**	0.60**	0.65**	0.35**
Lakes without reduction (*n*=20)	0.91**	0.89**	0.91**	0.88**
Lakes with reduction (*n*=8)	0.88*	0.68	0.82	0.37

In contrast to 'all lakes' and 'lakes with reduction', addition of the internal loading or a variable representing the sediment composition results only in a minor improvement of the correlation for lakes without reduction of the external loading. Replacing internal P loading by P_{sed} or the P_{sed}/Fe_{sed} ratio never improves the relation, but still a good significant correlation appears between the measured and calculated concentration in the lakes. Again it turns out that the P/Fe ratio gives better results than only the P content.

Discussion

The external P loading and the hydraulic retention time dominate the P concentration in lakes where no significant reduction of the external sources has been carried out. The 'classical' relations between external loading and concentration are useful for these lakes to describe the P concentration. Addition of the measured internal loading or the sediment composition in this type of relations did not improve the predictive value of the relation. Probably, lakes without reduction of the external loading have an equilibrium between water and sediment and consequently, addition of the internal loading to the relation will not improve the prediction of the P concentration in the lake. Therefore, it is expected that deviations of the concentration predicted by the Vollenweider model from the measured concentration for lakes without reduction of the external loading are largely due to errors in determining the loading to the lake. Another explaining factor might be differences in trophic structure between lake systems, but we did not gather sufficient data in our study to prove this statement.

In contrast, for lakes after reduction of the external P loading, it is not the external loading, but the internal P loading that is important in explaining the variations in the lake P concentration. Before reduction of the external loading the external and internal loading are correlated, but after a decrease in external loading no relation could be found. Thus, it is not always clear how the internal loading reacts to a decrease of the external loading and one should be careful using data on internal loading before the measures for prediction of the lake concentration after reduction.

Because of the more 'conservative' character of the sediment composition, the use of parameters representing its composition are preferred to the use of measured internal loading. Besides, the P and Fe content of the sediment can be determined relatively easily. Both the internal loading and the determination of the sediment composition are hampered by spatial heterogeneity, but for the latter low values are more accurate to determine. Low P concentrations in the lake water, important when the effect on algal growth is considered, correspond with low internal loadings and P contents in the sediment. Another advantage of using sediment P content instead of measuring the P release, is the possibility of setting up a mass balance of phosphorus over the top sediment, e.g. as conceptually formulated by Lijklema (1986). Although some serious methodological problems in setting up the balances have to be tackled (Smits & Van der Molen, 1993), mass balances over the sediment might be helpful in studying the extent and duration of the contribution of sediment P to the overlying water.

The P/Fe ratio is a better measure for the amount of P available for the lake concentration than the total P content or a combination of P fractions according to the scheme of Hieltjes & Lijklema (1980). Correlations between measured and calculated summer P concentrations were only slightly less when the P/Fe ratio was used instead of the measured internal loading. Jensen *et al.* (1992) found for Danish lakes also significant correlations between surface sediment P/Fe and mean values of the P concentration in the lake in summer and in winter. As expected, the concentration in the lake water decreased with decreasing values of the P/Fe ratio. The relatively good correlation between the P/Fe ratio and the concentration in the lake is caused by the higher values of this ratio. Possibly, the good correlation of higher values of the ratio with the P concentration in the lake can be explained as follows: higher values of the ratio represent a situation in which the adsorption complex is supersaturated and the 'extra' P is strongly available and therefore correlated with P in the lake water.

Direct relations between sediment composition and internal loading are generally weak. Jensen *et al.* (1992) found that the P/Fe ratio explained 58% of the variation in the measured internal loading, but in contradiction to the results of our study, ratios above 0.1 gave poor correlations. Higher values for the ratio were caused by a low sediment Fe content. They concluded that in these lakes organic P pools controlled the rate of P release. Boström (1984) reported a lack of correlation due to dominating amounts of non-releasable P.

The best results in the correlation between sediment composition and measured internal loading were obtained for the sum of the loosely bound and iron and aluminum bound P according to the fractionation scheme of Hieltjes & Lijklema (1980). Jensen *et al.* (1992) found a significant correlation between the loosely bound P and the measured internal P loading. Probably, during the measurements of the release of P under laboratory conditions the loosely bound P is used and released, but no freshly formed organic P settles and fills this pool again. Consequently this turnover is important for the measured fluxes, but has a minor nett effect on the lake P concentration. For anoxic sediments Nürnberg (1988) found good correlations between the measured internal loading and both total and reductant-soluble P in the sediment, while e.g. Shaw & Prepas (1990) found no significant correlation for shallow lake sediments.

Conclusions

External P loading and the hydraulic retention time are the main variables in determining the P concentration in lakes without reduction of the external loading. Internal loading or sediment composition do not significantly improve the relation. The broad confidence limits in the 'classical' relations must therefore be explained by uncertainty in the estimation of the loading, rather than effects of the internal loading.

The internal loading is the dominating factor in explaining the variation in the P concentration of lakes after a reduction in external loading. The external loading is of minor importance. This means that the widely used relations of Vollenweider (1975, 1979) and Vollenweider & Kerekes (1982) cannot be used to predict the lake concentration after a decreased external loading.

It is uncertain how the internal loading will react to a decrease of the external loading. Sediment composition, with respect to eutrophication, is less variable in time and easier to determine than internal loading. Especially the P/Fe ratio in the sediment is a good substitute for the internal loading in describing the P concentration in lakes after reduction of the external loading. Thus before reduction of the P loading, the best prediction of the lake concentration after the reduction can be based on the P/Fe ratio in the sediment and the expected changes in external P (and hydraulic) loading. Because the number of lakes available for the data analysis is rather small, it is necessary to improve the relation with more data.

When using more variables than external P loading and hydraulic loading, multiple regression relations are preferred to the Vollenweider type of relations.

Appendix

Data used in the calculations

s	with (x) or without (o) reduction of the external loading
D	depth (m)
L_{ext}	external phosphorus loading (gP m^{-2} y^{-1})
L_{int}	internal phosphorus loading (gP m^{-2} y^{-1})
σ	hydraulic retention time (y)
P_{lake}	phosphorus concentration in the lake (gP m^{-3})
P_{sed}	phosphorus content of the sediment (gP kg^{-1} dry weight)
Fe_{sed}	iron content of the sediment (gFe kg^{-1} dry weight)
Ref	Reference (refers to the end of the appendix)

		s	D	L_{ext}	L_{int}	σ	P_{lake}	P_{sed}	Fe_{sed}	Ref
1	Alserio	o			1.3			0.73	4.3	a
2	Apeldoorns Kanaal	o	1.6		4.4		0.83	10.72	39.4	
3	Arreskov	o			2.8			1.33	14.0	
4	Balaton Kes	o	3.2	2.5	0.2	0.28	0.10	0.78	16.3	b, c, d, e
5	Balaton Tihany	o	3.2		0.0	2.08	0.03	0.66	15.3	b, c, d, e
6	Bergse Plassen	o	2.0	20.3	11.0	0.30	1.50	9.21	26.9	
7	Bergundasjön 1	o	2.3	6.60	14.2	0.58	1.16	5.9	043.0	f, g, h, i
8	Braassemermeer	o	3.5	10.3	9.1	0.2	20.4	61.8	025.3	j
9	Brandermeer	o	1.3	6.93	1.5	0.08	0.19		30.9	
10	Breukeleveense Plas	o	1.5	1.27	0.3	0.19	0.11	1.00	15.0	
11	Brielse Meer	o	5.5	5.96	10.8	0.31	0.24	2.53	25.7	
12	Eemmeer	o	1.4	29.3	14.4	0.08	1.18			
13	Elfhoeven	o	3.0	1.71	5.1	0.50	0.16	1.00	20.0	j
14	Fysingen	o	2.0	0.54	0.3	0.50	0.05	1.43	43.0	g, i, k
15	Geerplas	o	1.6		4.2		0.52	1.20	10.5	l
16	Gooimeer	o	4.1	12.7	2.3	0.28	0.67	1.43		
17	Groot Vogelenzang	o	1.5		3.0		0.22	0.89	44.7	m
18	IJsselmeer N	o	4.4	3.10	6.4	0.42	0.27	1.06	8.3	
19	IJsselmeer Z	o	4.0	27.9	12.4	0.08	0.30	0.26	3.9	
20	Kagerplassen	o	2.8	21.5	21.9	0.09	0.69	2.38	14.4	
21	Kievitsbuurt Z	o			1.5			3.50	49.4	
22	Kortenhoefse Plas	o	1.2		8.0		0.23	1.30	10.0	m
23	Kvind	o			9.1			5.03	29.0	
24	Satoftasjön	o			1.8			2.07	39.0	k
25	Trummen 1	x	1.0		5.3		0.80	4.50		n
26	Trummen 2	x	1.4		0.4		0.09	1.46		n
27	Lillesjön	x	2.0	0.25	3.3	0.25	0.28	1.63	21.0	
28	Loosdrechtse Plassen	x	1.8	0.26	0.5	0.70	0.12	1.00	20.0	o, p
29	Markermeer	o	3.3	1.51	1.3	0.81	0.13	0.63	13.0	
30	Meerplas	o	1.6	0.00	2.9	1.82	0.12	0.69		
31	Naardermeer	o	1.0	0.42	1.5	0.53	0.11	1.88	42.1	
32	Nieuwkoop N	o	2.8	0.48	0.2	1.75	0.12	1.48	21.3	m
33	Nieuwkoop Z	o	2.8	4.70	5.7	0.33	0.31	2.28	17.6	m
34	Norrviken	x	5.4	0.45	2.2	0.80	0.12	1.76	33.0	g, q, r
35	Poel 't Zwet	o	1.0		12.6		0.52	1.84	25.9	
36	Slotermeer	o	1.6	1.34	0.6	0.46	0.18	0.48	4.2	
37	Søbygard	x	1.0	5.00	47.5	0.07	0.80	8.50	50.0	r, s
38	Spaarbekken Andijk	o	3.0	12.0	14.6	0.06	0.29	2.10		
39	Tjeukemeer	o	1.7	1.50	0.9	0.51	0.22	0.26	6.9	
40	Trehorningen 1	x	2.0	0.74	2.2	0.32	0.24			t
41	Trehorningen 2	x	2.0	0.70	4.4	0.47	0.44			t
42	Vaeng	x	1.2	1.50	6.8	0.05	0.13	4.80	63.0	
43	Vallentunasjön	x	2.7	0.14	4.0	1.67	0.30	1.81	33.0	g, i, k
44	Veluwemeer	x	1.3	1.12	1.4	0.39	0.14	0.36	16.1	u
45	Vuntus	o	1.5	0.24	0.4		0.09	0.60	14.0	
46	Waalboezem	o	3.8	6.21	5.0	0.31	0.17	0.62	10.7	
47	Wolderwijd	o	1.6	0.39	1.7	2.13	0.26	0.22	4.9	
48	Zwartemeer	o	1.0		1.5		0.33	0.88		
49	Bergundasjön 2	x	2.3	0.40	14.2	1.00	0.67	5.90	43.0	f, g, h, i

References				
	a	Premazzi & Provini (1985)	l	Klapwijk & Bruning (1986)
	b	Istvanovics (1988)	m	Van der Does *et al.* (1992)
	c	Herodek (1984)	n	Bengtsson *et al.* (1975)
	d	Herodek *et al.* (1988)	o	Boers *et al.* (1984)
	e	Pettersson & Istvanovics (1988)	p	Boers & Van Hese (1988)
	f	Bengtsson (1975)	q	Ahlgren (1980)
	g	Ryding (1981)	r	Sas (1989)
	h	Boström (1984)	s	Søndergaard (1987)
	i	Boström (1988)	t	Ryding (1982)
	j	Klapwijk *et al.* (1982)	u	Jagtman *et al.* (1992)
	k	Boström & Petterson (1982)		

References

Ahlgren, I., 1980. A dilution model applied to a system of shallow eutrophic lakes after diversion of sewage effluents. *Archiv für Hydrobiologie* 89: 17-32.

Bengtsson, L., 1975. Phosphorus release from a highly eutrophic lake sediment. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 19: 1107-1116.

Bengtsson, L., S. Fleischer, G. Lindmark & W. Ripl, 1975. Lake Trummen restoration project I. Water and sediment chemistry. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 19: 1080-1087.

Boers, P.C.M., J.W.Th. Bongers, A.G. Wisselo & Th.E. Cappenberg, 1984. Loosdrecht Lakes restoration project: sediment phosphorus distribution and release from the sediments. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 22: 842-847.

Boers, P.C.M. & D.T. Van der Molen, 1992. Invloed van de fosfaatnalevering op de fosfaatconcentratie in meren voor en na sanering van de externe belasting. RIZA nota 92.021, Lelystad (in Dutch).

Boers, P.C.M. & O. Van Hese, 1988. Phosphorus release from the peaty sediments of the Loosdrecht Lakes (The Netherlands). *Water Research* 22: 355-363.

Boström, B., 1984. Potential mobility of phosphorus in different types of lake sediments. *Internationale Revue der gesamten Hydrobiologie* 69: 457-474.

Boström, B., 1988. Relations between chemistry, biomass and activity in sediments of a sewage-polluted vs a nonpolluted lake. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 23: 451-459.

Boström, B. & K. Petterson, 1982. Different patterns of phosphorus release from lake sediments in laboratory experiments. *Hydrobiologia* 92: 415-429.

Cooke, G.D., E.B. Welch, S.A. Peterson & P.R. Newroth (eds.), 1986. Lake and reservoir restoration. Ann Arbor Science, Butterworth Publishers.

Emerson, S., 1976. Early diagenesis in anaerobic lake sediments: chemical equilibria in interstitial waters. *Geochimica et Cosmochimica Acta* 40: 925-934.

Herodek, S., 1984. The eutrophication of lake Balaton: Measurements, modelling and management. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 22: 1087- 1091.

Herodek, S., V. Istvanovics & J. Zlinszky, 1988. Phosphorus metabolism and eutrophication control of lake Balaton. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 23: 517-521.

Hieltjes, A.H.M. & L. Lijklema, 1980. Fractionation of inorganic phosphates in calcareous sediments. *Journal of Environmental Quality* 9: 405-407.

Holdren, G.C. & D.E. Armstrong, 1980. Factors affecting phosphorus release from intact lake sediment cores. *Water Science and Technology* 14: 79-87.

Hosper, S.H., 1989. Biomanipulation, new perspective for restoring shallow, eutrophic lakes in the Netherlands. *Hydrobiological Bulletin* 23: 5-10.

Hosper, S.H. & M-L. Meijer, 1986. Control of phosphorus loading and flushing as restoration methods for Lake Veluwe, the Netherlands. *Hydrobiological Bulletin* 20: 183-194.

Istvanovics, V., 1988. Seasonal variation of phosphorus release from the sediments of shallow lake Balaton (Hungary). *Water Research* 22: 1473-1481.

Jagtman, E., D.T. Van der Molen & S. Vermij, 1992. The influence of flushing on nutrient dynamics, composition and densities of algae and transparency in Veluwemeer, The Netherlands. *Hydrobiologia* 233: 187-196.

Jensen, H.S., P. Kristensen, E. Jeppesen & A. Skytthe, 1992. Iron:phosphorus ratio in surface sediment as an indicator of phosphate release from aerobic sediments in shallow lakes. *Hydrobiologia* 235/236: 731-743.

Jeppesen, E., P. Kristensen, J.P. Jensen, M. Søndergaard, E. Mortensen & T. Lauridsen, 1991. Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Memorie dell 'Istituto Italiano di Idrobiologie* 48: 127-148.

Klapwijk, S.P., J.M.W. Kroon & M.-L. Meijer, 1982. Available phosphorus in lake sediments in The Netherlands. *Hydrobiologia* 92: 491-500.

Klapwijk, S.P. & C. Bruning, 1986. Available phosphorus in the sediments of eight lakes in The Netherlands. In: P.G. Sly (ed.), Sediment/Water Interactions. Springer Verlag, New York, pp. 391-398.

Larsen, D.P. & H.T. Mercier, 1976. Phosphorus retention capacity of lakes. *Journal of the Fisheries Research Board of Canada* 33: 1742-1750.

Lijklema, L., 1986. Phosphorus accumulation in sediments and internal loading. *Hydrobiological Bulletin* 20: 213-224.

Moss, B., H. Balls, K. Irvine & J. Stansfield, 1986. Restoration of two lowland lakes by isolation from nutrient-rich water sources with and without removal of sediment. *Journal of Applied Ecology* 23: 391-414.

Nürnberg, G.K., 1988. Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. *Caadian Journal of Fisheies and Aquatic Sciences* 45: 453-462.

Pettersson, K. & V. Istvanovics, 1988. Sediment phosphorus in lake Balaton-forms and mobility. *Archiv für Hydrobiologie* 30: 25-41.

Premazzi, G. & A. Provini, 1985. Internal P loading in lakes: a different approach to its evaluation. *Hydrobiologia* 120: 23-33.

Reckhow, K.H. & S.C. Chapra, 1979. A note on error analysis for a phosphorus retention model. *Water Resources Research* 15: 1643-1646.

Ryding, S.-O., 1981. Reversibility of man-induced eutrophication. Experiences of a lake recovery study in Sweden. *Internationale Revue der gesamten Hydrobiologie* 66: 449-503.

Ryding, S.-O, 1982. Lake Trehorningen restoration project. Changes in water quality after sediment dredging. *Hydrobiologia* 92: 549-588.

Sas, H., 1989 (ed.). Lake restoration by reduction of nutrient loading: expectations, experiences and extrapolations. Academia Verlag Richarz GmbH, St. Augustin, Germany.

Shaw, J.F.H. & E.E. Prepas, 1990. Relationships between phosphorus in shallow sediments and in the trophic zone of seven Alberta lakes. *Water Research* 24: 551-566.

Smits, J.G.C. & D.T. Van der Molen, 1993. Application of SWITCH, a model for sediment-water exchange of nutrients, to Lake Veluwe in The Netherlands. *Hydrobiologia* 253: 281-300.

Søndergaard, M., 1987. Phosphorus release from a hypertrophic lake sediment: experiments with intact sediment cores in a continuous flow system. *Archiv für Hydrobiologie* 116: 45-59.

Van der Does, J. P.J.T. Verstraelen, P.C.M. Boers, J. Van Roestel, R.M.H. Roijackers & G. Moser, 1992. Lake restoration with and without dredging of phosphorus-enriched upper sediment layers. *Hydrobiologia* 233: 197-210.

Vollenweider, R., 1975. Input-output models, with special reference to the phosphorus loading concept in limnology. *Schweizerische Zeitschrift für Hydrologie* 37: 53-84.

Vollenweider, R., 1979. Das Nährstoffbelastungskonzept als Grundlage für den externen Eingriff in den Eutrophierungsprozess stehender Gewässer und Talsperren. *Zeitschrift für Wasser- und Abwasser-Forschung* 12: 46-56.

Vollenweider, R. & J. Kerekes, 1982. Eutrophication of waters, monitoring, assessment and control. OECD, Paris.

Williams, J.D.H., J.K. Syers, S.S. Shukla, R.F. Harris & D.E. Armstrong, 1971. Levels of inorganic and total phosphorus in lake sediments as related to other sediment parameters. *Environmental Science and Technology* 5: 1113-1120.

Mathematical modelling as a tool for management in eutrophication control of shallow lakes

D.T. Van der Molen[1], F.J. Los[2], L. Van Ballegooijen[1] & M.P. Van der Vat[2], 1994

In: E. Mortensen, E. Jeppesen, M. Søndergaard & L.K. Nielsen (eds.), Nutrient Dynamics and Biological Structure in Shallow Freshwater and Brackish Lakes. Proceedings International Conference, 16-20 August 1992, Silkeborg, Denmark. *Hydrobiologia* 275/276: 479-492.

1 Institute for Inland Water Management and Waste Water Treatment, Lelystad, The Netherlands
2 Delft Hydraulics, Delft, The Netherlands

Abstract

The eutrophication model DELWAQ-BLOOM-SWITCH was developed to be a functional tool for water management. Therefore it includes nutrients, algal biomass and composition as well as water transparency. A module describing the interaction between water and bottom gives the model the flexibility to deal with measures, such as a decrease of the external phosphorus loading and flushing with water differing in composition from the lake water. This paper focuses on the functional aspects of the model, the results of an application on Lake Veluwe, The Netherlands, and the implications for water management.

With one set of coefficients DBS reproduces the most important characteristics of Lake Veluwe for a period of two years before measures (reduction of the external loading and flushing during the winter months) and eight years after the measures. The phosphorus concentration decreased and became growth limiting for algae instead of nitrogen and light. Both in measurements and modelling results, the algal composition changed from blue-green algae dominance to green algae and diatom dominance. Lake Veluwe had a relatively short transient phase after reduction of external loading, because high nitrate concentrations in the flushing water inhibited a long period with high phosphorus releases from the bottom.

Model calculations were carried out to investigate the effects of fish stock management and optimisation of flushing. Both measures are promising.

Introduction

Eutrophication is one of the main topics in water quality management of freshwater lakes since the last few decades. An excessive amount of nutrients causes abundant algal growth, which in turn reduces the natural and economical value of lakes. Several measures are taken to combat eutrophication (e.g. Cullen & Forsberg, 1988; Boers & Van der Molen, 1993), but only in a few cases the desired quality targets are achieved. After a reduction of the external loading, the internal loading from the bottom is able to supply algae with phosphorus until it reaches equilibrium with the lowered external loading (Lijklema, 1986; Sas, 1989; Van der Molen & Boers, 1993); according to a survey of 27 Danish lakes this may take 4 -16 years (Jeppesen *et al.*, 1991). Therefore, additional measures are introduced. Examples are dredging or chemical treatment of sediments (e.g. Björk, 1985; Cooke *et al.*, 1986; Boers *et al.*, 1992) and fish stock management (e.g. Benndorf, 1990).

Eutrophication is a complicated problem with several interrelated aspects. Important factors are nutrient levels, biomass and composition of algae and water transparency. This makes it difficult for a water quality manager to choose among alternative management scenarios to improve the water quality. A sound decision requires a great deal of knowledge concerning the system. In addition to measures, application of models provides a possible way of dealing with these complicated systems.

Simple, empirical models (e.g. Vollenweider, 1975) and expert opinions can give an indication of the results that may be expected from measures, but have serious shortcomings for understanding lake ecosystems (Van der Molen & Boers, 1993). Hence, the earliest operational deterministic eutrophication models were already developed in the early seventies (DiToro *et al.*, 1971). The transient phase of a lake after reduction of the external loading can only be described by dynamical models, with inclusion of the nutrient dynamics in the bottom. Besides nutrients, algal biomass and species composition as well as water transparency are constraints for a management eutrophication model. Considering the significance of all these pieces of information, the relations between them and the necessity to create an operational tool for water managers, we have developed the DBS (DELWAQ-BLOOM-SWITCH) model system.

In this paper we present the model. Specifying all model equations and process coefficients would require too much space and is beyond the scope of this paper. Therefore we shall only mention the main processes and coefficients and focus on the results of application and the possibilities offered by DBS. We will illustrate this using the application of the model to Lake Veluwe, The Netherlands, as an example. Lake Veluwe is a shallow, eutrophic lake in which from 1979 onwards several measures have been taken. Optimisation of the current measures and application of additional measures are points of interest for water managers. Relevant data are available from before and after the measures, making the construction of reliable input possible as well as validation of the results.

Modelling approach

General set-up

DBS includes modules for physical-, (bio)chemical- and biological processes (Los, 1992, 1993). DBS is one of several modelling systems using the general DELWAQ framework (Postma, 1988). DELWAQ serves four major purposes:

- it contains the physical schematisation,
- it calculates transport of substances as a function of advective and dispersive transport, processes and loads,
- it accumulates fluxes and computes resulting concentrations for each time-step; a large number of numerical solution schemes are included in DELWAQ and can be selected by the modeller constructing a DELWAQ based model,
- it produces output in a standardised way, which can easily be processed further, creating graphs and statistics.

The actual water system is represented within DBS by means of one or several segments. Segments are computational units, which can be considered homogenous with respect to the processes included in the model. Segments can be arranged 0-, 1-, 2- or 3 dimensionally. Water transports between segments must, however, be known in advance. For complicated systems with many segments these transports might be derived from dedicated models. For simple (lake) systems the user might simply specify the water flows to and from the system. Internally DBS multiplies fluxes with concentrations to obtain masses across internal and external boundaries.

Several modules have been developed and tested previously. Computations concerning phytoplankton are performed by the BLOOM II model, which has been shown to be accurate and reliable in both fresh and salt water systems (Los *et al.*, 1984; Los & Brinkman, 1988; Los, 1991). Included within DBS is also a light module which computes the components of the total extinction caused by non-living suspended matter as well as the Secchi depth (transparency) of the water

(Buiteveld, 1990). The contribution of phytoplankton to the total extinction is computed by BLOOM II. The module that describes the interactions between water and bottom (SWITCH) was developed specifically for DBS, although it was also tested in a stand alone version (Smits & Van der Molen, 1993). To operate the model system we have simultaneously developed a 'user-friendly' interface.

DBS creates a detailed report for each individual segment containing all concentrations plus additional information (i.e. on limiting factors, light regime, depth, etc.) and the size of each flux through the system. Output can be inspected in the form of graphs and tables.

Kinetic modules

DBS considers the cycles of carbon, oxygen and three nutrients: nitrogen, phosphorus and silicon. Each of these elements can appear in several different compartments:

- inorganic nutrients (nitrogen and phosphorus are subdivided into several fractions of inorganic nutrients),
- dead organic material (two groups of detritus),
- phytoplankton (several groups),
- grazers (zooplankton and mussels),
- bottom algae (diatoms),
- detritus at the bottom - water interface (we call this 'complex' detritus as many organic, inorganic, biochemical and biological processes occur in a complicated way within this thin boundary layer),
- bottom detritus,
- inorganic bottom nutrients.

With the exception of grazers the amount in each compartment is computed by DBS. Often this is done by solving a series of differential equations numerically. Several equations, however, are solved analytically, some processes are assumed to be at steady state and the phytoplankton production is calculated using an optimisation technique. Chemical, physical and biological processes result in fluxes from one compartment to another. The basic cycles for the three nutrients are identical. The most important fluxes within DBS and some additional information are presented in (Figure 1).

DBS considers three main phytoplankton groups: diatoms, green algae and blue-green algae. Each of these groups is further differentiated into types, according to their growth-limitation. To compute the amount of each type, an optimisation technique is used to maximise the total net production of all types in a certain period of time, consistent with the environmental conditions and the existing biomass levels. The environmental conditions include water temperature, solar radiation, depth, background extinction and the concentrations of available nutrients. Each type in the model is characterised by a different set of physiological parameters such as growth and mortality rates and nutrient requirements. The model selects the optimal composition of phytoplankton species, based upon their physiological characteristics and the available resources.

Although formulations are available to model zooplankton (e.g. Jørgensen, 1983), results are rarely validated with observations. Therefore, the biomass of grazers is an input to DBS. It can be provided in the form of observed (or any other series of) numbers or in the form of a function. The model does check whether the computed biomass of phytoplankton and various forms of detritus can sustain the amount of grazers specified in the input. If not, it will reduce the amount of grazers

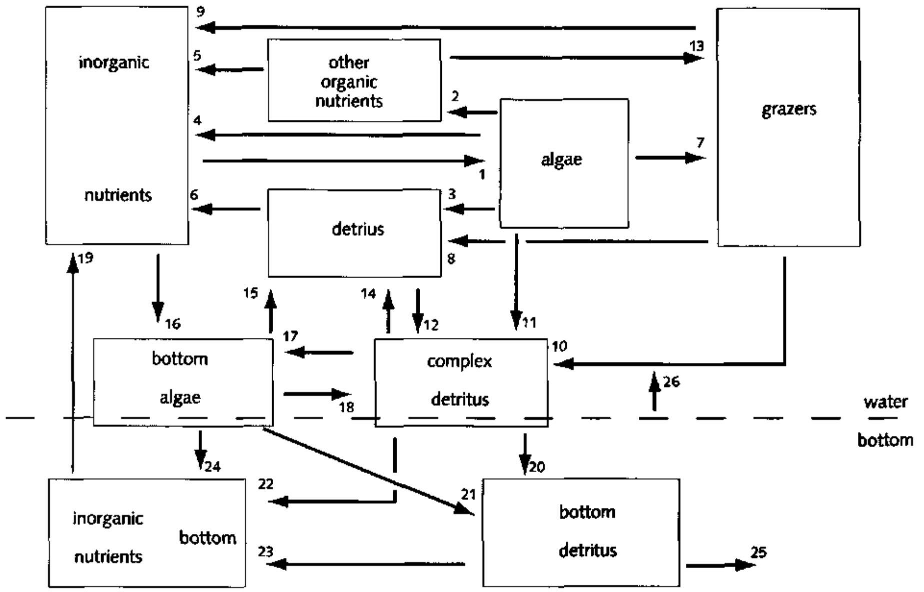


Figure 1 *The main compartments and fluxes in DBS; numbers refer to:*

1. Uptake of inorganic nutrients by phytoplankton. DBS assumes that ammonia is (completely) depleted first before the phytoplankton switch to nitrate.
2. Phytoplankton mortality flux to other organic substances.
3. Phytoplankton mortality flux to detritus.
4. Phytoplankton mortality flux to inorganic nutrients (autolysis). The sum of fluxes 2+3+4 equals the natural phytoplankton mortality.
5. Mineralisation of other organic substances. It is assumed that this process is very slow and that it only takes place in the water column.
6. Mineralisation of detritus. The actual rate depends not only on temperature, but also on the composition of detritus. It is maximal when there are sufficient nutrients relative to carbon.
7. Mortality of phytoplankton due to grazing.
8. Production of detritus due to zooplankton grazing (faeces).
9. Release of inorganic nutrients by grazers.
10. Production of complex detritus due to grazing by mussels (faeces).
11. Sedimentation of phytoplankton.
12. Sedimentation of detritus in the water phase.
13. Sedimentation of other organic nutrients, which become refractory.
14. Resuspension of complex detritus.
15. Resuspension of bottom phytoplankton.
16. Uptake of inorganic nutrients by bottom algae. This process is activated if process 17 cannot supply enough nutrients.
17. Uptake by bottom algae of dissolved nutrients released through mineralisation of complex detritus in the water-bottom interface.
18. Mortality flux of bottom algae to complex detritus.
19. Total flux of inorganic bottom nutrients to the water. This flux is influenced by all processes included within the bottom module of DBS.
20. Burial of complex detritus.
21. Burial of bottom algae.
22. Mineralisation of complex detritus. For conceptual reasons this flux is handled by the bottom module; nutrients are not directly released into the surface water.
23. Mineralisation of bottom detritus.
24. Part of bottom algae mortality flux to inorganic nutrients (autolysis).
25. Burial of bottom detritus. This process actually removes nutrients from the system.
26. Sediment growth. This results in dilution of the contents in sediment; usually it is set zero.

to the highest possible level that can be supported by the system.

Two types of dead organic matter are distinguished, detritus and a pool named 'other organic nutrients'. The mineralisation of both types of detritus results in the regeneration of nutrients. Mineralisation is described as a first order process and the mineralisation rate is dependent on the nutrient to carbon ratio of the material. Higher ratio's result in increased mineralisation rates. 'Other organic nutrients' mineralise very slowly and sedimentation and burial of this fraction are considered as removal of nutrients from the system.

For phosphorus in the overlying water we have included adsorption and desorption of phosphate, as well as degradation and sedimentation of adsorbed phosphorus. Nitrification and denitrification are assumed to take place in the bottom only.

The bottom is divided into an 'active' and an 'inactive' layer. The active layer is usually assumed to be 0.1 m and is divided into four layers: the aerobic layer, the denitrifying layer, the upper reduced layer and the lower reduced layer. Layers can change in thickness according to the circumstances; the thickness of the aerobic layer and the denitrifying layer are derived from the differential equations of oxygen and nitrate in the bottom, respectively. The mineralisation rate of bottom detritus is lower for the reduced layers than for the oxidised layer. Nitrate is formed in the oxidised layer by first order nitrification of ammonium. The removal of nitrogen by denitrification of nitrate in the denitrifying layer is also described as a first order process. For phosphorus this bottom module includes adsorption and desorption processes (according to the Langmuir isotherm) and precipitation and dissolution of phosphorus minerals (first order kinetics with the difference between the actual concentration and the saturation concentration as a driving force). These processes also vary over the four bottom layers. Hence the phosphorus flux to the surface water varies significantly as a function of the redox situation in the bottom.

Model application

Case study Lake Veluwe

Lake Veluwe (The Netherlands) is an artificial lake which was formed when the Flevoland polder was created in 1956 (Hosper & Meijer, 1986). The lake has a total surface area of $32.8 \cdot 10^6$ m^2; approximately half of the lake is very shallow (depth less than 1 m) with a sandy sediment; the other half is deeper (average depth 2 m) with a more silty sediment (Figure 2). The water is assumed to be well mixed, so the system is modelled as one segment.

In the second half of the sixties the lake quality deteriorated, because of increased nutrient loading. This resulted in high chlorophyll *a* concentrations and an almost permanent bloom of blue-green algae (predominantly *Oscillatoria agardhii)* from 1970 onwards. In 1979 the following restoration measures were taken: from February onwards external phosphorus loading was reduced from 3 to 1.0 - 1.5 gP m^{-2} y^{-1} by phosphorus elimination at the sewage treatment plant discharging its effluent to the lake. Furthermore, flushing the lake with water from the Flevoland polder (poor in algae and phosphorus, but rich in calcium and nitrate) during the winter, decreased the retention time of dissolved compounds from 0.35 to 0.15 years. In the second half of the eighties, the lake was flushed in summer as well, decreasing the retention time of dissolved compounds from 0.50 to 0.25 years (Jagtman *et al.*, 1992).

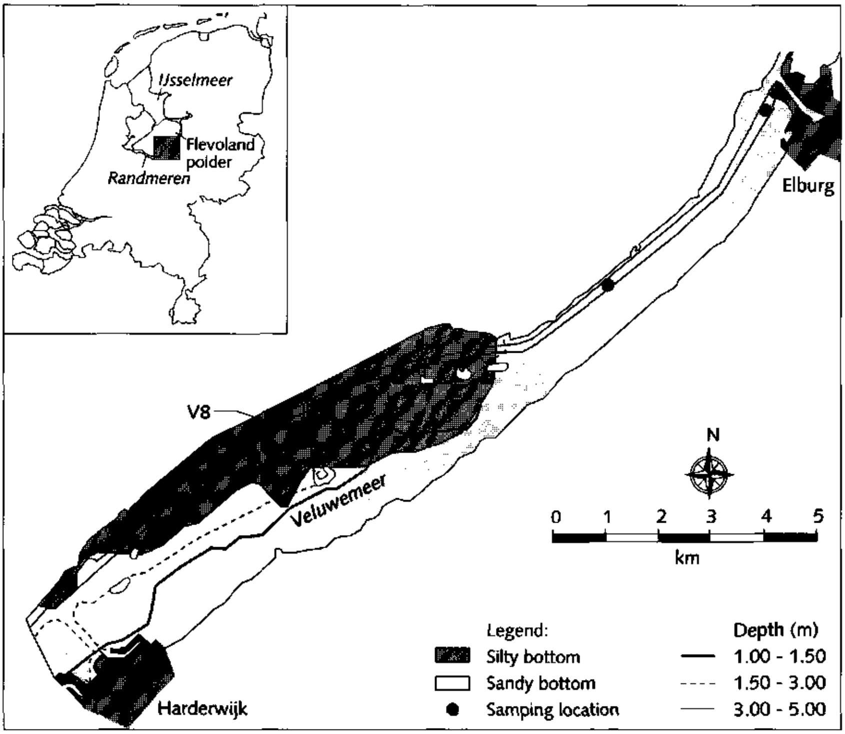


Figure 2 *Lake Veluwe.*

The performance of DBS was checked for the Lake Veluwe case for the years 1978 through 1987. As no data on loadings were available for 1984 and from 1988 onwards, we ran the model separately for two periods: 1978 through 1983 and 1985 through 1987. The initial conditions for 1978 were obtained from measurements, the initial conditions for 1985 were taken from the computations at the end of 1983. We used a single set of model coefficients for the entire period of time. So, effectively DBS was applied to the entire ten year period.

Almost all coefficients for the water phase had been obtained during previous studies for many different lakes, including Lake Veluwe. These studies include laboratory and field work (Zevenboom *et al.*, 1982; Brinkman & Van Raaphorst, 1986) as well as model studies in which similar formulations were used (Los *et al.*, 1984; Los & Brinkman, 1988; Los, 1991). For the recently developed bottom module minor modifications of the stand-alone calibration/-validation, as reported by Smits & Van der Molen (1993), were necessary. As this paper focuses on the results of the model application and the implications for management, only the values of the most important coefficients are given here (Table 1).

Table 1 *Values of some important coefficients of the model* DBS *(rates at 20 °C; DW = dry weight, PW = pore water).*

Coefficient	Value/range	Unit
Growth rate algae	0.68 - 1.36	d^{-1}
Mortality rate algae	0.16 - 0.23	d^{-1}
Respiration rate algae	0.05 - 0.12	d^{-1}
C:P ratio algae	42.1 - 88.9	$g\ g^{-1}$
N:P ratio algae	3.64 - 8.00	$g\ g^{-1}$
C:chlorophyll ratio algae	25.0 - 50.0	$g\ g^{-1}$
Mineralization rate detritus C, N and P	0.12 - 0.18	d^{-1}
Sedimentation rate detritus	0.05	$m\ d^{-1}$
Maximum filtration rate zooplankton	1.5	$m^3\ g^{-1}C\ d^{-1}$
Diffusion coefficients O_2, PO_4, NO_3, NH_4	$4.20 - 9.33 \cdot 10^{-5}$	$m^2\ d^{-1}$
Adsorption capacity bottom	0.4 - 0.8	$gP\ kg^{-1}DW$
Ortho P saturation concentration bottom	0.05	$gP\ m^{-3}PW$
Mineralisation rate oxidised bottom	0.03	d^{-1}
Nitrification rate bottom	50.0	d^{-1}
Denitrification rate bottom	50.0	d^{-1}

To illustrate the possibilities this model offers for management purposes, two scenario's were studied in addition to the reference case: fish stock management and optimisation of flushing with polder water. Fish stock management might be promising, because the phosphorus loading and concentration is sufficiently low to expect success of this intervention (Benndorf, 1990; Jeppesen *et al.*, 1990). Reconsideration of flushing is necessary, because from 1985 onwards the phosphorus concentration in the lake approached the concentration in the flushing water. Optimisation was established by flushing only when the phosphorus concentration in the other inlet water was significantly higher than the concentration in the flushing water. The water level was maintained.

Results

In the two years (1978 and 1979) prior to the onset of the sanitation measures both computed and observed levels of chlorophyll range between 100 mg m^{-3} in winter and about 400 mg m^{-3} in summer (Figure 3). There is a significant drop in 1980 and a more gradual decline during the remainder of the 1980s. During the last year (1987) computed and observed chlorophyll levels are well below 100 mg m^{-3} all year long. The agreement between computed and observed concentrations varies to some extent. For example computed levels for 1983 fall below those for 1982, but the opposite is true in the measurements. Still it may be concluded that the model does reproduce the trend over this ten year period of time sufficiently well.

In addition to biomass both the observed and computed (Figure 4) phytoplankton species composition changed dramatically during this ten year period. Before 1980 blue-green algae, mainly *Oscillatoria agardhii*, dominated all year long, except for a short period of diatom dominance in spring. In the early summer of 1982 green algae replaced the blue-greens for the first time, but blue-green algae returned later and remained dominant in 1983 and 1984. The computed result for 1983, suggesting a summer dominance of green algae, is incorrect. From 1985 to 1987, however, diatoms

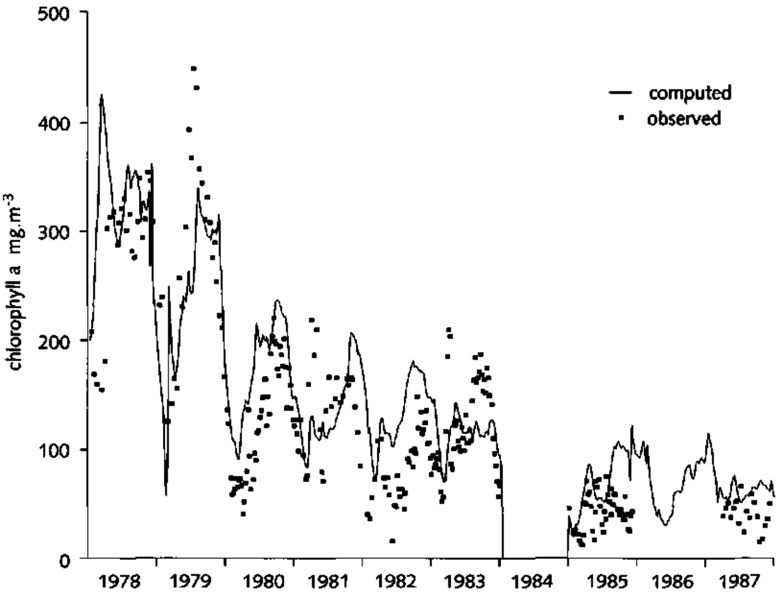


Figure 3 *Computed and measured chlorophyll in Lake Veluwe.*

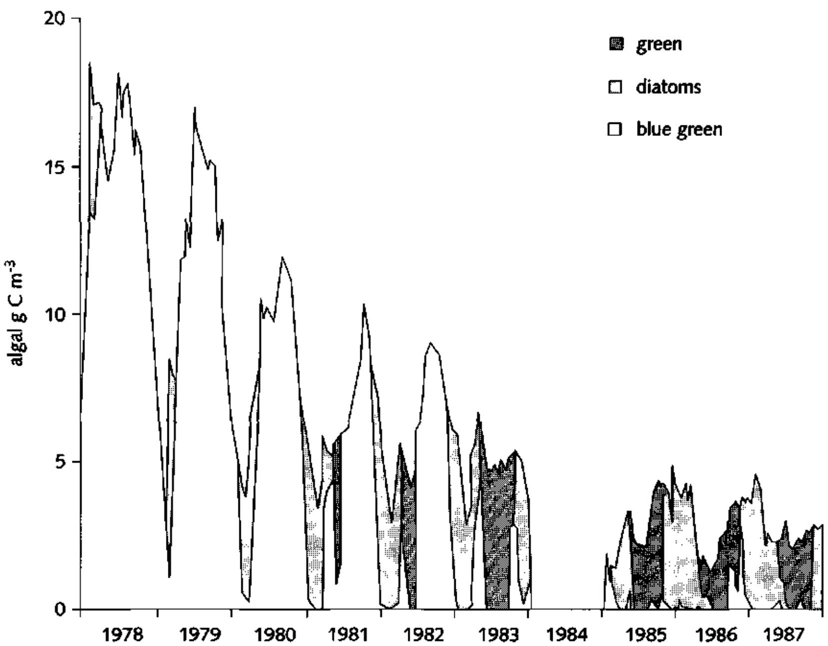


Figure 4 *Computed algal composition in Lake Veluwe.*

and green algae were dominant and blue-green algae (re)appeared infrequently. For two representative years (1985 and 1986) the computed species are presented with measurements (Figure 5). The model does not reproduce the reappearance of diatoms in autumn 1985 correctly. However, in general the computed species composition agrees well with the observed trend.

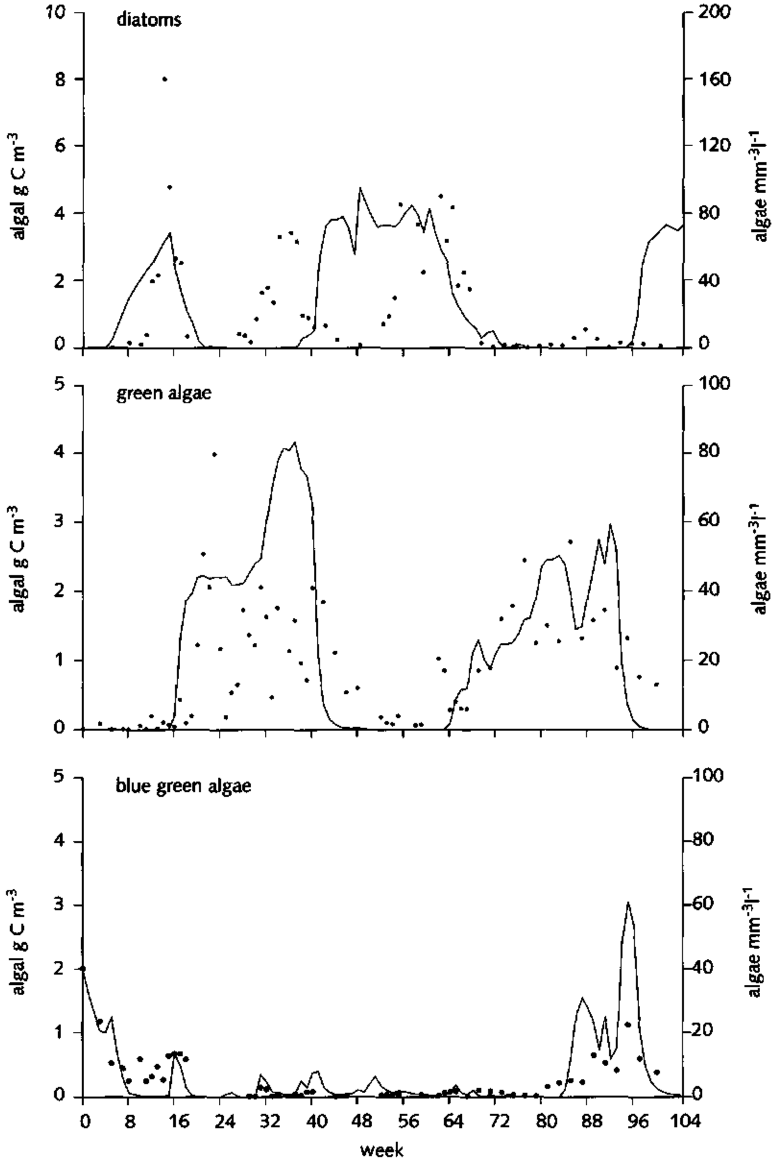


Figure 5 *Computed (left axis; line) and observed (right axis; dots) algal species in Lake Veluwe for 1985 - 1986.*

As a consequence of the improved light conditions near the bottom, bottom algae increased dramatically from 1985 onwards. The model computes that from 1985 to 1987 almost 20% of the algal carbon can be found on the bottom. Measurements based on chlorophyll indicate that in the shallow part this can be more than 50% (Van der Molen & Helmerhorst, 1991), meaning that model result is probably an underestimation.

The decrease in algal biomass is mainly due to a dramatic decrease in the amount of phosphorus in the water phase (Figure 6). In 1978 and 1979 the available amount of phosphorus exceeded the requirements of phytoplankton. Algal growth was usually limited by lack of nitrogen (summer) and light energy. Starting in 1980, the levels of phosphorus became progressively more restraining to the phytoplankton community, although nitrogen limitations still occurred in the summer. During the last three years when phosphorus levels decreased to about 0.1 g m^{-3}, phosphorus was in fact the only important limiting factor. Observed and computed levels of ortho phosphorus (not shown here) were mostly equal to zero. As for chlorophyll, the model results agree sufficiently well with the observations. The largest deviation occurs in 1980: the year immediately following the onset of the sanitation measures. The real system reacted even more dramatically than the model in this transient year.

The phosphorus content of the bottom changed only slowly, but varied within the limits of the measurements (not shown). However, sediment data are scarce and difficult to interpret (Smits & Van der Molen, 1993).

Computed levels of nitrate (Figure 7) and ammonia (Figure 8) agreed well with observations. Notice that the period of nitrate depletion during summer became shorter during this ten year period, indicating that nitrogen became progressively less important as a limiting factor. As was the

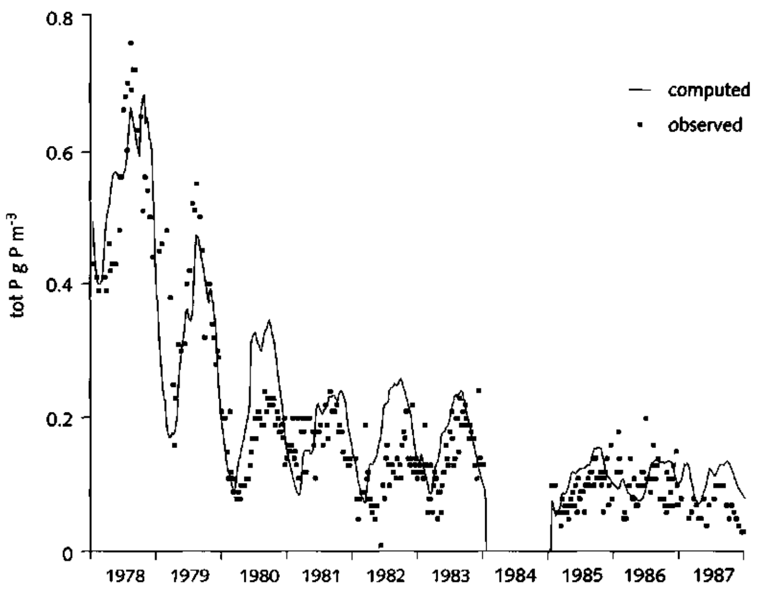


Figure 6 *Computed and measured total phosphorus in Lake Veluwe.*

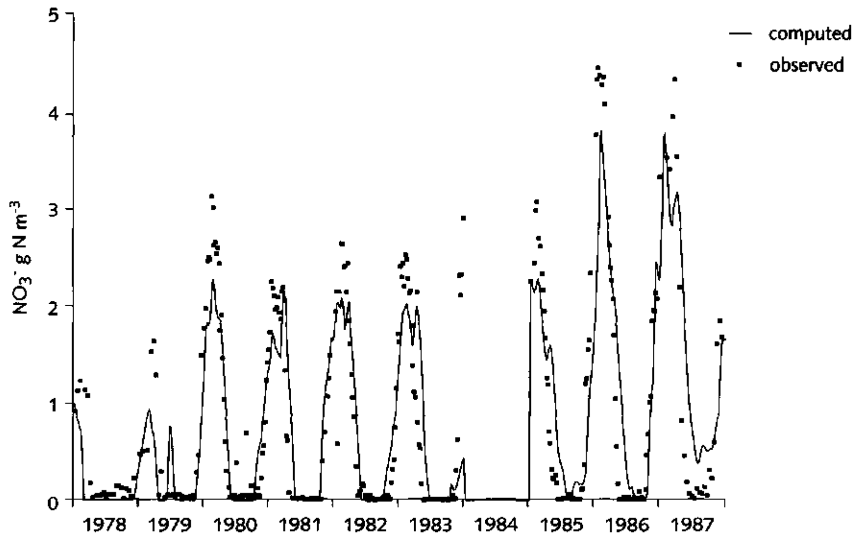


Figure 7 *Computed and measured nitrate in Lake Veluwe.*

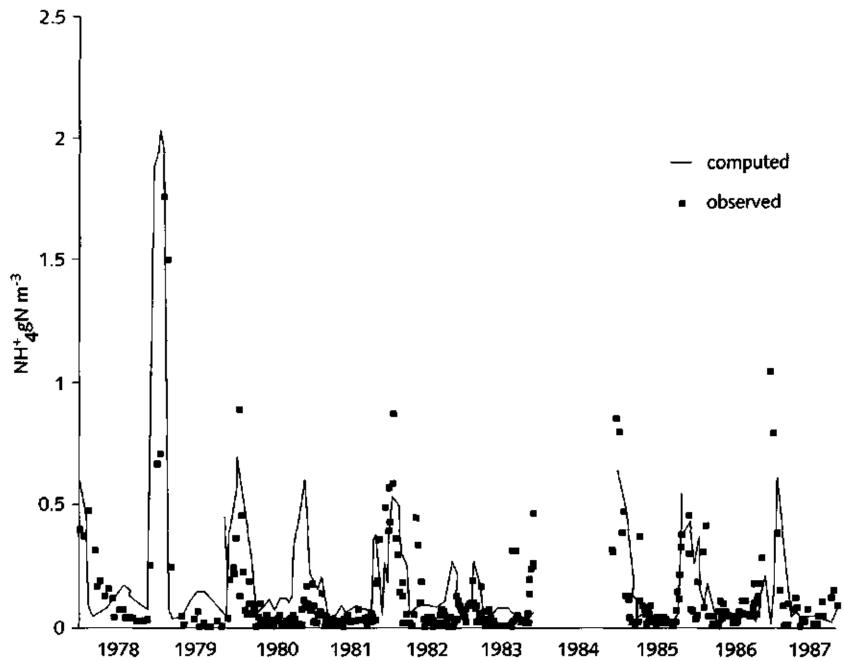


Figure 8 *Computed and measured ammonium in Lake Veluwe.*

case for phosphorus there was also a decreasing trend in the computed and observed values for Kjeldahl nitrogen (total nitrogen minus nitrate) during this ten year period, though less pronounced (Figure 9). This can be explained from the fact that some, but not all fractions of Kjeldahl nitrogen declined significantly. In later years less nitrogen was present as a component of phytoplankton and detritus, but the amounts in 'other organic nitrogen' and ammonia generally remained constant. Before the measures, denitrification was limited by nitrate production from nitrification in the sediment. As a result of the flushing of Lake Veluwe with nitrate-rich water, nitrate became an important fraction of total nitrogen. From 1980 onwards the model computed increased denitrification as a result of the enhanced nitrate diffusion into the sediment.

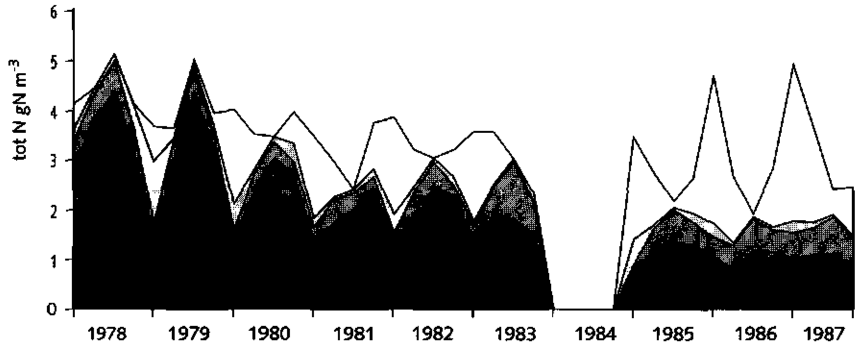


Figure 9 *Computed nitrogen fractions in Lake Veluwe (three monthly averaged).*

In the results of the model application we have seen that DBS is able to reproduce some important characteristics of the system. Similar or even better agreement with measurements exists for other substances such as silicon and oxygen. An accurate reproduction of concentrations can, however, also be the result of compensating errors in individual fluxes. This is a general problem with models: many concentrations can be validated, but few fluxes can. Moreover, most of these processes cannot be measured very accurately. For 1983 and the period from 1987 onwards, infrequent measurements of the phosphorus release from the bottom were available (Figure 10).

The phosphorus release was measured under laboratory conditions using the continuous flow system with the temperature ranging from 10 - 21 °C (Brinkman & Van Raaphorst, 1986; Boers & Van Hese, 1988). The measurements are in the same range as the model results, although the maximum value in 1983 is not reproduced. Both measurements and simulation show a decrease in release rates. The calculated sediment oxygen demand is also comparable with measurements in 1983 and 1987 (not shown).

The effect of fish stock management was simulated by increasing the zooplankton biomass. Although the zooplankton biomass was increased by a factor of 4, the effective increase of zooplankton grazing was only a factor of 2.5, due to food limitation. No preference for certain algal species was included. Zooplankton biomass compares well with data from Boers *et al.* (1991), although after removal of 75% of the fish stock in Lake Wolderwijd, The Netherlands, no increase of the grazing capacity was observed (Meijer *et al.*, 1994). The optimised flushing resulted in a

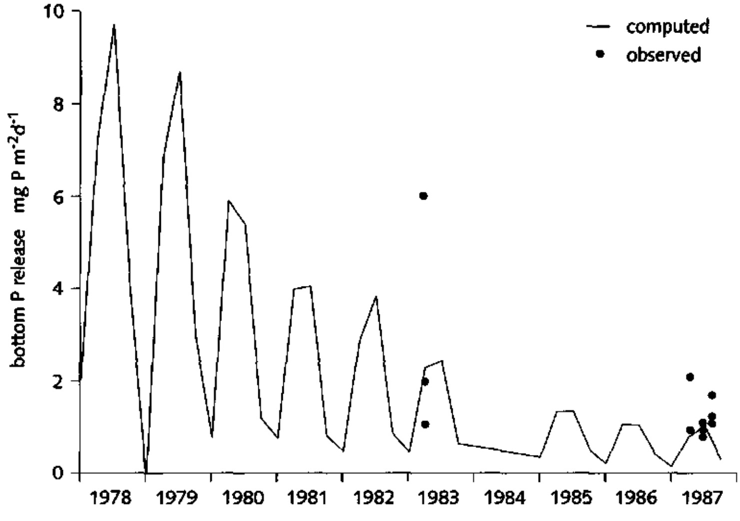


Figure 10 *Computed (three monthly averaged) and measured phosphorus release from the bottom in Lake Veluwe.*

significant decrease of the total yearly water load, although there was an increase in the spring.

The results of the simulation of fish stock management showed that both nitrogen and phosphorus concentrations were lower when compared to the reference calculations and, consequently, chlorophyll was decreased and transparency increased. Biomass of blue-green algae diminished as a consequence of the increased grazing. About 75% of algal bound carbon was present in bottom algae. This compares well with measurements in a comparable lake after fish stock management (Van der Molen & Helmerhorst, 1991).

The results from the optimised flushing regime calculations indicate the importance of the phosphorus concentration in the inlet water. Dilution with the inlet water in periods when the phosphorus concentration was high in the lake decreased the total algal biomass, but the percentage of blue-green algae increased slightly. The percentage of bottom algae did not change.

Discussion

An important phenomenon is the change in algal species dominance from blue-greens to greens and diatoms. According to the model, this is not a direct but an indirect result of declining phosphorus levels. Blue-green algae require less phosphorus per unit of biomass than green algae, so one might expect them to remain dominant as phosphorus becomes increasingly limiting. The decrease in biomass, however, also results in improved light conditions and hence in higher potential growth rates of all species and of green algae and diatoms in particular (due to the form of their growth versus light relationship). Between 1982 and 1985 the growth rates of green algae and diatoms become sufficiently high to displace the blue-green algae, but competition was intense. The additional reduction in available phosphorus and hence biomass in 1985 - 1987 was sufficient to provide greens and diatoms a clear advantage over blue-greens. Still, the balance between blue-green algae on one hand and green algae and diatoms on the other is unstable. Initial and climatic conditions, e.g. ice coverage, were also found to be important (Jagtman *et al.*, 1992).

The biomass of bottom algae is possibly underestimated, although their increasing importance is reproduced in the model results. Probably the improved light conditions near the bottom and relatively high availability of nutrients are favourable for this group of algae.

The decrease of algal biomass in the water phase results in better light conditions. On the other hand, green algae and diatoms have a higher turn-over rate and produce detritus which degrades more slowly, hence an increase of detritus can be expected (Gunnison & Alexander, 1975). The net effect of increased detritus concentration and decreased chlorophyll concentrations was a minor increase in water transparency (Figure 11).

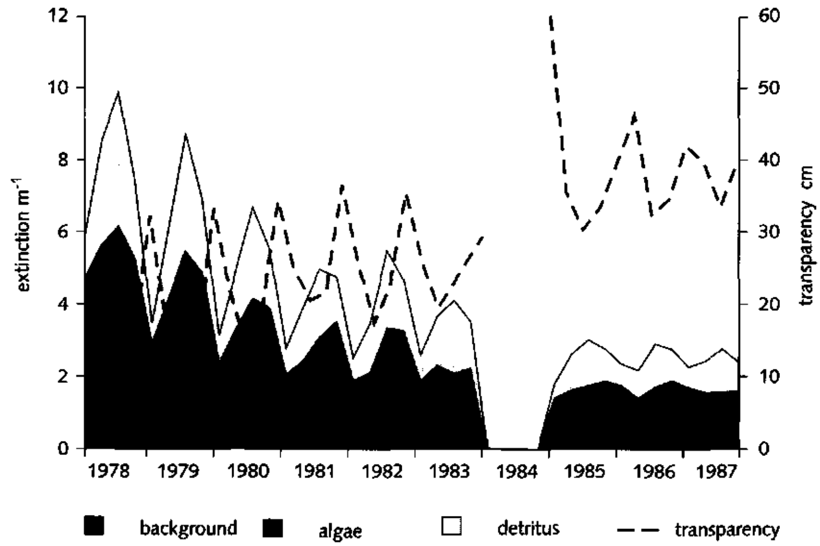


Figure 11 *Computed extinction (cumulative) and transparency in Lake Veluwe (three monthly averaged).*

As already mentioned, the reduction of the concentration of available phosphorus is the main cause for the decline of phytoplankton biomass. In Table 2 we distinguish the main processes within the water phase and fluxes across the bottom - water interface. In 1978 the external loading was high, just as in the previous years. Losses of phosphorus to the bottom exceeded the release from the bottom: the bottom became a sink for phosphorus. The retention of phosphorus (quotient of the external loading minus outflow and the external loading) was approximately 55%. The system began to change in 1979. The external loadings to the lake declined and the amount of phosphorus in the water decreased significantly. Interesting enough, the net exchange with the bottom was small: yearly averaged retention was less than 10%, while in summer the retention became negative due to a relatively high internal loading. The organic flux to the bottom was still high, but inorganic sedimentation was much lower than in 1978.

Table 2. *Main processes (yearly averaged) of the phosphorus cycle (direct losses from the bottom such as seepage and burial to deeper layers are omitted).*

Process rate ($mgP\ m^{-2}\ d^{-1}$)	Time period			
	1978	1979	1980 - 1983	1985 - 1987
External loading	8.7	4.5	4.2	4.2
Outflow	-3.9	-4.1	-3.0	-2.5
To bottom (inorg)	-4.1	-0.8	-0.2	-0.2
To bottom (org)	-7.1	-5.0	-3.4	-2.2
From bottom	5.8	4.6	2.3	0.8
Total water	-0.6	-0.8	-0.1	+0.1
Total bottom	-5.4	-1.2	-1.3	-1.6

In the period 1980 through 1983 the retention increased to 40%. Notice, that the inorganic sedimentation of phosphorus was less than 10% of the organic sedimentation. The trends for the 1980 - 1983 period continued during the 1985 - 1987 period. The phosphorus release from the bottom decreased by a factor of 7 compared to 1978 and was also small compared to the external loading. Since the inorganic phosphorus sedimentation during these periods was insignificant, the bottom processes were dominated by the mineralisation of settled and buried detritus. The computed retention was about 40%. This is less than the value derived from the computation with mass balance equations, because the phosphorus concentration and consequently the outflow were slightly overestimated.

The phosphorus release from the bottom decreased more than was proportional to the external loading. The model computes an increase of the thickness of the oxidised layer as a result of the decreased sedimentation of organic matter and the bottom uptake of nitrate from the overlying water. So, as a consequence of the flushing, the recovery of Lake Veluwe took place in a few years, which is short when compared to many other lakes (e.g. Cooke *et al.*, 1986; Sas, 1989; Jeppesen *et al.*, 1991). Therefore, additional measures to reduce the release of phosphorus from the bottom, such as dredging and chemical fixation of phosphorus, do not seem useful in the case of Lake Veluwe.

More likely, effects may be expected from a further decrease of the external phosphorus loading, since phosphorus is the growth limiting factor. Two additional measures are examined in this paper: fish stock management and optimisation of the flushing rate. Increased grazing, assumed to be the result of the fish stock management, resulted in a lowered algal biomass and consequently in a decreased detritus concentration. This caused a remarkable increase of transparency. Transparency was almost doubled and was limited by particular, inorganic matter. As this is not included in the model, we assumed that this was not influenced by the measures. The increased algal turnover resulted in increased organic sedimentation, which was favourable for the bottom algae. However, these results are partly hypothetical. Meijer *et al.* (1994) showed that a 75% removal of the fish stock in the large and shallow neighbouring Lake Wolderwijd resulted in a significant decrease in primary production, while grazing only increased slightly. Meijer *et al.* (1994) suggest that a high zooplankton biomass could not sustain due to predation by the mysid shrimp *Neomysis integer* and that the lowered primary production was due to reduced internal loading. This clearly illustrates the fact that food web interactions are not yet fully understood and that including such interactions in mathematical models must be done with caution.

Flushing lost its diluting effect to a large extent, but concentration of flushing in periods with relatively high phosphorus concentrations in the other inlet water had positive effects. Both chlorophyll and total phosphorus decreased. Due to the lowered total flushing rate, the percentage of blue-greens increased, because a high water retention time is advantageous to blue-green algae with their low growth and mortality rates. Total nitrogen decreased, as a result of reduced nitrate loading. Low nitrate concentrations resulted in a less oxidised state of the sediment, but due to the lowered organic sedimentation, no increase of the phosphorus release from the bottom was computed. Organic sedimentation was low in the reference calculations when compared to the sedimentation calculated in the simulation of fish stock management, resulting in nutrient limitations of bottom algae. Only 20% of total algal biomass consisted of bottom algae.

Conclusions

The eutrophication model DBS is capable of reproducing the most important water quality indicators. With one set of coefficients the model was able to simulate a 10 year period, in which the external phosphorus loading of Lake Veluwe decreased significantly and flushing was applied. The model calculates concentrations in the bottom and fluxes through the system which are comparable to the available measured data. The coefficients were derived from previous laboratory, field and model studies, except for some bottom coefficients.

The measures taken resulted in decreased algal biomass as well as in a change in the algal composition from blue-green algae dominance towards green algae and diatom dominance. Flushing lowered the phosphorus concentration, but increased the nitrate concentration in the lake. According to the model results, phosphorus became the growth limiting factor instead of nitrogen and light energy.

After the measures the phosphorus release from the bottom strongly decreased, more so than the external loading. Apart from the lowered input of organic material, high nitrate concentrations in the flushing water also caused the oxidised layer of the bottom to become thicker and reduced internal loading. This implies that dredging and phosphorus fixation in the bottom cannot be expected to have much effect.

Therefore calculations were made to study the effects of fish stock management and optimisation of the flushing. If an increase of the grazing capacity by a factor of 2 - 3 can be realised, dissolved nutrient levels, as well as chlorophyll and detritus concentrations can be lowered significantly. The transparency will increase by nearly a factor of 2. However, reduction of the fish stock in a large lake will probably not always result in an increase of grazing capacity and the chain of events caused by fish stock management measures is still not fully understood. The effect of flushing only when the phosphorus concentration in the inlet water is high, is also clear, though less spectacular. Due to low organic sedimentation it is expected that the lowered total flushing will not impede further improvements of Lake Veluwe, through increased internal loading. These examples point out that DBS can be helpful in understanding the processes in lake ecosystems and that the model is a powerful tool for reviewing various management strategies.

References

Benndorf, J., 1990. Conditions for effective biomanipulation; conclusions derived from whole-lake experiments in Europe. *Hydrobiologia* 200/201: 187-203.

Björk, S., 1985. Lake restoration techniques. In: Proceedings International Congress Lake Pollution and Recovery, Rome, pp. 281-292.

Boers, P.C.M., L. Van Ballegooijen & E.J.B. Uunk, 1991. Changes in phosphorus cycling in a shallow lake due to food web manipulations. *Freshwater Biology* 25: 9-20.

Boers, P.C.M., J. Van der Does, M. Quaak, J. Van der Vlugt & P. Walker, 1992. Fixation of phosphorus in lake sediments using iron(III)chloride: experiences, expectations. *Hydrobiologia* 233: 211-212.

Boers, P.C.M. & D.T. Van der Molen, 1993. Eutrophication control in Europe; state of the art. *European Water Pollution Control* 3: 19-25.

Boers, P.C.M. & O. Van Hese, 1988. Phosphorus release from the peaty sediments of the Loosdrecht Lakes (The Netherlands). *Water Research* 22: 355-363.

Brinkman, A.G. & W. Van Raaphorst, 1986. De fosfaathuishouding in het Veluwemeer. Proefschrift Technische Hogeschool Twente, The Netherlands (in Dutch with an English summary).

Buiteveld, H., 1990. UITZICHT - model voor berekening van doorzicht en extinktie. RIZA nota 90.058, Lelystad (in Dutch).

Cooke, G.D., E.B. Welch, S.A. Peterson & P.R. Newroth (eds.), 1986. Lake and reservoir restoration. Ann Arbor Science, Butterworth Publishers.

Cullen, P. & C. Forsberg, 1988. Experiences with reducing point sources of phosphorus to lakes. *Hydrobiologia* 170: 321-336.

DiToro, D.M., J.J. Fitzpatrick & R.V. Thoman, 1971. A Dynamic Model of the Phytoplankton Population in the Sacramento San Joaquim Delta. *Adv. Chem. Ser.* 106: 131-180. American Chemical Society, Washington D.C.

Gunnison, D. & M. Alexander, 1975. Resistance and susceptibility of algae to decomposition by natural microbial communities. *Limnology and Oceanography* 20: 64-70.

Hosper, S.H. & M-L. Meijer, 1986. Control of phosphorus loading and flushing as restoration methods for Lake Veluwe, The Netherlands. *Hydrobiological Bulletin* 20: 183-194.

Jagtman, E., D.T. Van der Molen & S. Vermij, 1992. The influence of flushing on nutrient dynamics, composition and densities of algae and transparency in Veluwemeer, The Netherlands. *Hydrobiologia* 233: 187-196.

Jeppesen, E., P. Kristensen, J.P. Jensen, M. Søndergaard, E. Mortensen & T. Lauridsen, 1991. Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Memorie dell 'Istituto Italiano di Idrobiologie* 48: 127-148.

Jeppesen, E., M. Søndergaard, E. Mortensen, P. Kristensen, B. Riemann, H.J. Jensen, J.P. Müller, O. Sortkjaer, J.P. Jensen, K. Christoffersen, S. Bosselmann & E. Dall, 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes. II Threshold levels, long-term stability and conclusions. *Hydrobiologia* 200/201: 219-227.

Jørgensen, S.E., 1983. Modeling the ecology processes. In: G.T. Orlob (ed.), Mathematical Modeling of Water Quality: Streams, Lakes and Reservoirs. IIASA-Wiley, New York, pp. 116-149.

Lijklema, L., 1986. Phosphorus accumulation in sediments and internal loading. *Hydrobiological Bulletin* 20: 213-224.

Los, F.J., 1991. Mathematical simulation of algae blooms by the model BLOOM II, Version 2, Documentation report. Delft Hydraulics, The Netherlands.

Los, F.J., 1992. Procesformuleringen DBS. T542, Waterloopkundig Laboratorium Delft, The Netherlands (in Dutch).

Los, F.J., 1993. Technisch rapport DBS. T542, Waterloopkundig Laboratorium Delft, The Netherlands (in Dutch).

Los, F.J. & J.J. Brinkman, 1988. Phytoplankton modeling by means of optimization: a 10-year experience with BLOOM II. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 23: 790-795.

Los, F.J., N.M. de Rooij & J.G.C. Smits, 1984. Modelling eutrophication in shallow Dutch Lakes. *Verhandlungen der Internationalen Vereinigung für Theorethische und Angewandte Limnologie* 22: 917-923.

Meijer, M.-L., E.H. Van Nes, E.H.R.R. Lammens, R.D. Gulati, M.P. Grimm, J. Backx, P. Hollebeek, E.M. Blaauw & A.W. Breukelaar, 1994. The consequence of a drastic fish stock reduction in the large and shallow Lake Wolderwijd, The Netherlands. Can we understand what happened? *Hydrobiologia* 275/276: 31-42.

Postma, L., 1988. DELWAQ Users manual, Version 3.0, Delft Hydraulics, The Netherlands.

Sas, H., 1989 (ed.). Lake restoration by reduction of nutrient loading: expectations, experiences and extrapolations. Academia Verlag Richarz GmbH, St. Augustin, Germany.

Smits, J.G.C. & D.T. Van der Molen, 1993. Application of SWITCH, a model for sediment-water exchange of nutrients, to Lake Veluwe in The Netherlands. *Hydrobiologia* 253: 281-300.

Van der Molen, D.T. & P.C.M. Boers, 1993. Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. *Hydrobiologia* 275/276: 379-390.

Van der Molen, D.T. & T.H. Helmerhorst, 1991. Bodemalgen in de randmeren. *H_2O* 24: 719-724 (in Dutch with an English summary).

Vollenweider, R., 1975. Input-output models, with special reference to the phosphorus loading concept in limnology. *Schweizerische Zeitschrift für Hydrologie* 37: 53-84.

Zevenboom, W., A.F. Post & L.R. Mur, 1982. Assessment of factors limiting growth rate of *Oscillatoria agardhii* in hypertrophic Lake Wolderwijd. *Limnology and Oceanography* 27: 39-52.

Changes in phosphorus and nitrogen cycling following food web manipulations in a shallow Dutch lake

D.T. Van der Molen[1] & P.C.M. Boers[1], 1996

Freshwater Biology 35: 189-202.

Institute for Inland Water Management and Waste Water Treatment, Lelystad, The Netherlands

Abstract

1. The flow of phosphorus and nitrogen through the food web of the shallow, eutrophic lake Wolderwijd was analysed for 2 different years before and for 1 year after food web manipulation.
2. After fish removal the water became clear and the growth of macrophytes began. Fish removal resulted in a significant reduction of the total nutrient pool in the water, but differences between the nutrient cycles before and after the experiment were mainly caused by a gradual change driven by a reduced phosphorus input.
3. The zooplankton biomass before and after food web manipulation did not chance significantly. Unfavourable food conditions and predation by young fish limited zooplankton biomass after the food web manipulation.
4. After fish removal benthic algae, fish, zoobenthos and macrophytes form the largest pools of nutrients apart from the sediment top layer. However, they contribute only little to nutrient cycles in the watercolumn.

Introduction

In lowland Europe eutrophication is still a major water quality problem and, while the problem has only recently been recognised in marine systems, several ameliorative measures have been applied to freshwaters, including the reduction of external nutrient loading (Boers & Van der Molen, 1993). Mostly, these measures are designed to decrease the nutrients available for phytoplankton, but food web manipulation is also used to decrease algal biomass. In The Netherlands, remarkable results have been obtained in several small lakes, in addition to some failures and some instances of only temporary effects (Meijer *et al.*, 1994a).

From November 1990 to July 1991, a large-scale food web manipulation project was carried out in Wolderwijd, a shallow lake of more than 2600 ha in the centre of The Netherlands. Seventy-five percent of the fish biomass was removed and a large number of young pike was introduced (Meijer *et al.*, 1994b). As part of the assessment of this initiative, Boers *et al.* (1991) estimated the fluxes of the internal phosphorus (P) cycle of the lake before and after the food web manipulation. They concluded that the increased grazing pressure would not only reduce phytoplankton, but also a decrease the availability of P by increasing the sedimentation of detrital P as a result of the growth of macrophytes. In this paper we evaluate the assumptions made by Boers *et al.* (1991) concerning the values of the parameters and the structure of the ecosystem.

After food web manipulation nitrogen (N) frequently becomes a limiting factor for algal growth if macrophytes cover a substantial part of the lake surface (Meijer *et al.*, 1994a). Macrophytes cause a decrease in the N concentration, directly, by uptake and, indirectly, by creating alternately aerobic and anaerobic zones in the sediment, enhancing denitrification (Caffrey & Kemp, 1992). By associating the P cycling with N we aim to understand more of the N dynamics after a food web manipulation experiment.

Study area

Wolderwijd (1918 ha, mean depth 1.5 m) is situated between the recently reclaimed (in 1968) polder South Flevoland and the original land (Figure 1). The lake has an open connection with Nuldernauw (6.97 ha, mean depth 1.65 m), which may be considered as a part of Wolderwijd. The main water transport direction is from north to south. Table 1 summarises the water, P and N budget of Wolderwijd. The hydrology of the lake is characterised by a higher water level in summer than in the winter. Occasionally large amounts of water are flushed through the lake from the neighbouring Lake Veluwe; the hydraulic load from Lake Veluwe varied from 0 $m^3\ y^{-1}$ in 1978 - 1979 to $108{\cdot}10^6\ m^3\ y^{-1}$ in 1992. Due to the use of phosphorus-free detergents and several restoration

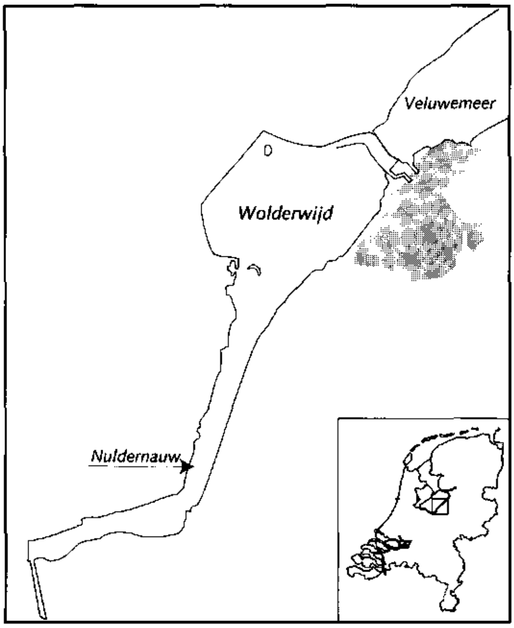


Figure 1 *The situation of Wolderwijd (inset is a map of The Netherlands).*

measures, the P concentration in the incoming water has decreased. This has resulted in a significant decrease in P, N and chlorophyll concentration in the lake over the past two decades (Figure 2). The percentage algal bound N has decreased, but nitrate-N has increased. Despite these changes Secchi depth and algal composition have improved only slightly. Secchi depth increased from 0.2 - 0.3 m (summer averaged) at the end of the seventies to 0.4 - 0.5 m in 1990. After the fish removal in 1991, Secchi depth increased to more than 1.0 m for a period of 6 weeks. In March and April 1991 the algal composition consisted of diatoms (*Diatoma* sp. and *Nitzschia* sp.) and green algae (*Scenedesmus* sp.). In summer cyanobacteria (*Oscillatoria* sp.) appeared and remained dominant in later years.

Table 1 *The water, P and N budget of Wolderwijd, The Netherlands, averaged over 1976 - 1992.*

	Water (10^6 m^3 y^{-1})	P (10^6 gP y^{-1})	N (10^6 gN y^{-1})
Inputs			
Precipitation	19.6	1.6	60.7
Dry deposition	0.0	No data	82.8
seepage	6.4	0.3	5.4
Polder	5.7	3.9	40.3
Streams	16.5	12.7	123
Load from Lake Veluwe	25.9	2.5	86.1
Outputs			
Net load to Lake Eem	-30.6	-16.7	-140
Evaporation	-18.3	0.0	0.0
Infiltration	-19.5	-2.8	-58.4

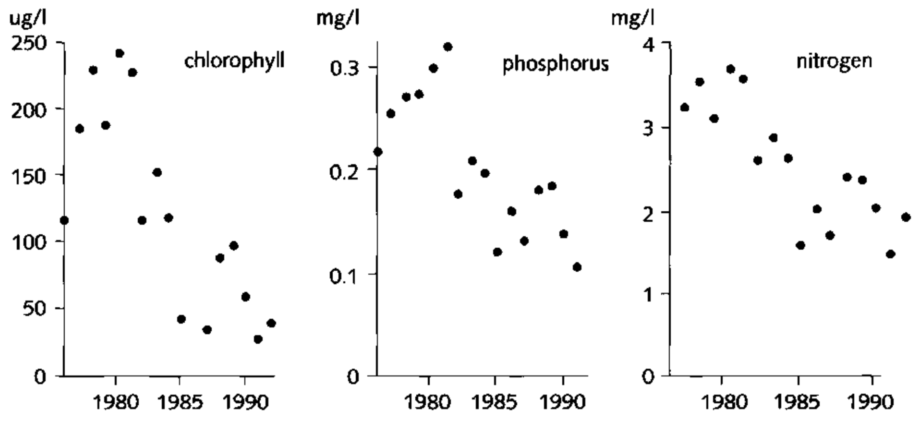


Figure 2 *Summer averaged chlorophyll (left), total phosphorus (middle) and total nitrogen (right) in Wolderwijd from 1976 to 1992.*

Materials and methods

Boers *et al.* (1991) analysed the flow of P through the food web of Wolderwijd for several years. The year 1981 represented the hypertrophic period, 1987 was chosen as a year without dominance of cyanobacteria, and a hypothetical year was used to predict the situation after food web manipulation. In 1991 fish were removed and the lake was intensively monitored. Therefore, we are now able to validate the assumptions, results and predictions of Boers *et al.* (1991) and to reconstruct the P cycle of Wolderwijd. Based on the P cycle, and additional ratios from data in the literature, we have also calculated an N cycle of the food web in Wolderwijd. Only data for the summer have been used. Figure 3 gives a schematic overview of the pools and fluxes which are distinguished in this study.

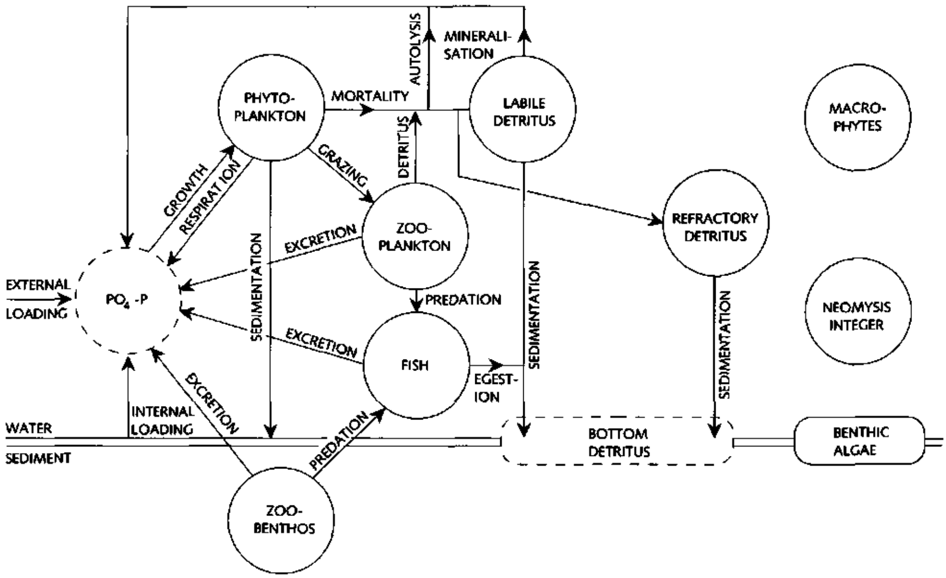


Figure 3 *Schematic representation of the pools and fluxes in the food web of Wolderwijd.*

External and internal nutrient loading

More accurate and distinctly different external P loadings to the estimates of Boers *et al.* (1991) have been derived from recently available mass balances, based on measurements of the hydraulic loadings and nutrient concentrations in the main inputs to the lake. Because the main external inputs are concentrated in winter, the water and nutrients loadings are relatively small in summer. The outflow of inorganic nutrients has been calculated by the product of the average measured concentration in summer and the calculated amount of fluid water that leaves the system during the summer.

The internal P loading was monitored in 1988, 1991 and 1992 and measured according to Boers & Van Hese (1988). In 1988 the average internal loading was 2.7 mgP m^{-3} d^{-1} in May and September at various locations. In 1991 the internal loading in summer (twenty-four data, five locations) decreased to 0.38 mgP m^{-3} d^{-1} (standard deviation 0.28). The internal P loading in the adjacent Lake Veluwe was also significantly lower in 1991 compared with previous years. In 1992 and 1993 both the internal loading and the standard deviation were almost a factor of two higher. Based on a linear relationship between the logarithm of summer total P concentration and the internal loading, we estimated the internal P loading in 1981 and 1987 to be 5.0 and 1.5 mgP m^{-3} d^{-1}, respectively. The internal loading of nitrate and ammonium was measured in the summer of 1993 and 1994. Measurements were carried out in light and dark conditions, to simulate the release with and without benthic algae (Van Luijn *et al.*, 1995). We used the data measured without light for 1981 and the data with light for 1991. The internal N loading for 1987 was derived from the data of 1981, assuming the same reduction as for the internal P loading. The measurements of the internal loading in the laboratory at 20 °C were corrected for the actual temperature of the lake water in summer using the Arrhenius equation.

A part of the settled organic material will mineralise in the sediment. The ammonium-N will partly transform to nitrate and denitrify to N_2 gas. The coupled nitrification-denitrification was measured in 1994 (Van Luijn *et al.*, 1995). The summer average value (21.5 mgN m^{-2} d^{-1}) was used for 1991, with the same correction for temperature as applied for the nutrient release rates. For 1981 and 1987 no data are available.

Phytoplankton

Chlorophyll was measured biweekly at two locations. Summer averaged chlorophyll was 232, 36 and 29 mg m^{-3} in 1981, 1987 and 1991, respectively, cyanobacteria contributing 100%, 30% and 52% of the total biovolume. Diatoms and green algae both made up 20-25% of the biovolume in 1991. The carbon : nutrient and carbon : chlorophyll ratios, summarised in Table 2 (Los, 1991), are derived from literature data. We assume that nutrients and carbon are taken up and lost in the same ratio as present in the algae. For 1981 and 1987, Boers *et al.* (1991) estimated growth rates as 0.18 d^{-1} and 0.54 d^{-1}, respectively, based on the changed algal composition and availability of light. In the summer of 1991 primary production was measured with the ^{14}C incubation method weekly at two locations. The average value was 0.69 gC m^{-3} d^{-1} (standard deviation 0.39). Bioassay experiments showed both P and N to limit algal growth in the summer of 1991.

Table 2 *Parameters for components in the food web of Wolderwijd, The Netherlands, averaged for the growth season.*

	Carbon, dry weight	Phosphorus	Nitrogen
Cyanobacteria	40(gC g^{-1} chlorophyll)	60 (gC g^{-1}P)	5 (gC g^{-1}N)
Other algae	33 (gC g^{-1} chlorophyll)	60 (gC g^{-1}P)	5 (gC g^{-1}N)
Zooplankton		42 (gC g^{-1}P)	4.5 (gC g^{-1}N)
Fish	25 (gDW g^{-1}WW)	2.7 (%DW)	10.6 (%DW)
Zoobenthos		1.0 (%DW)	10.0 (%DW)
Macrophytes		0.22 (%DW)	1.75 (%DW)

Respiration rates, 0.05 d^{-1} for cyanobacteria and 0.2 d^{-1} for other algae, were derived from Zevenboom (1980). Physical processes, such as flushing and sedimentation, contributed only little in the removal of nutrients in algae. The flushing rate in summer is low (<0.001 d^{-1}), and, therefore a net sedimentation of only 0.03 d^{-1} (Scavia, 1980) was used. Mortality was calculated as the difference between growth and the sum of respiration, grazing and sedimentation.

Zooplankton

In the summer of 1991 the average measured biomass was 0.36 gC m^{-3}, with maximum values up to 1.0 gC m^{-3} in the period mid-May to mid-June. Most biomass was made up of *Daphnia hyalina* Leydig and *D. galeata* (Sars). In previous years *Daphnia* sp. and *Bosmina longirostris* Müller were both present in comparable numbers, but the biomass was still dominated by the larger *Daphnia*. Also, the time of collapse of the zooplankton peak (June) did not differ from previous years.

Several grazing parameters were measured during the summer of 1991 every 2 weeks in two locations. The specific ingestion rate, measured according Gulati *et al.* (1982) and corrected according to Boers *et al.* (1991), was 0.44 d^{-1}. This is considerably lower than the values assumed for

1981 and 1987 (1.1 d^{-1} and 0.9 d^{-1}, respectively; Boers *et al.*, 1991).

The average measured ingestion rate was 0.627 gC m^{-3} d^{-1} and the average measured assimilation rate 0.210 gC m^{-3} d^{-1}. The assimilation efficiency calculated for 1991 was 40.4%, the quotient of measured assimilation and ingestion. For the previous years an assimilation efficiency of 40% was used (Boers *et al.*, 1991). The fraction not assimilated is assumed to become detritus. Respiration of zooplankton was assumed to be 10% - 15% (average 13%) of biomass d^{-1} (Gulati *et al.*, 1982). Further, we assumed that the nutrient excretion is related to the carbon respiration according to the carbon : nutrient ratio in zooplankton (Table 2; Gulati *et al.*, 1982; Sterner *et al.*, 1992). Growth of zooplankton was calculated as the difference between the assimilated fraction of the material grazed and the sum of excretion by the zooplankton and predation by fish.

Detritus

In our model, detritus in the water column is formed by mortality of phytoplankton and egestion by zooplankton. We assumed that 35% of the nutrients are directly released by autolysis. The remaining part is divided into a labile and a stable fraction, based on the phytoplankton composition. We assumed that 90% of the detritus of cyanobacteria (Gunnison & Alexander, 1975) and 50% of the detritus of other phytoplankton species (Jewell & McCarthy, 1971) is labile. The detritus concentration was calculated from the steady state equation, in which production is equal to the sum of mineralisation and removal by sedimentation. A mineralisation of 0.5 d^{-1} is used for labile detritus and zero for stabile detritus, so the production equals the sedimentation. For 1981 and 1987 we assumed a net sedimentation of 0.06 d^{-1}. Boers *et al.* (1991) used a sedimentation of 0.12 d^{-1}, but they did not take sedimentation of phytoplankton into account.

Boers *et al.* (1991) assumed there would be a considerable increase in the rate of the removal by sedimentation after fish removal, because they expected an increase in submerged macrophytes. Indeed, clear water is frequently observed within the parts of the lake with abundant macrophyte coverage. The biomass of macrophytes actually did increase (see 'Macrophytes' below), but the area occupied was almost constant. We assumed a smaller effect on the net sedimentation than did Boers *et al.* (1991) and we included sedimentation of phytoplankton, so for 1991 we assumed a removal of 0.10 d^{-1} instead of 0.30 d^{-1}.

Fish

The fish stock before the food web manipulation was estimated at 300 kg fresh weight ha^{-1} in 1981 (Boers *et al.*, 1991) and 203 kg fresh weight ha^{-1} in 1987 (Meijer *et al.*, 1994a). The most important species in terms of biomass were bream (*Abramis brama* L.) and roach (*Rutilis rutilis* L.). Because 157.4 kg ha^{-1} fish was removed in winter 1990 - 1991, 45.6 kg ha^{-1} remained: about 25 kg benthivorous fish and 20 kg planktivorous fish ha^{-1}. However, measurements indicated that from June to September 1991 the fish biomass had increased to 115 kg ha^{-1}. Fish migration was prevented, so net fish production in summer 1991 was 69.4 kg ha^{-1}. Almost half of this increase was caused by ruffe (*Gymnocephalus cernua* L.). The average fish biomass of 1991 was assumed to be the sum of the starting biomass and 50% of the net production. Only 8% of the introduced pike (*Esox lucius* L.) survived, resulting in a standing stock of predatory fish of 5 - 10 kg ha^{-1}. The diet of the young pike consisted only of zooplankton (Meijer *et al.*, 1994b). The conversion factors to dry weight and to nutrient contents are derived from Penczak & Tatrai (1985) (Table 2).

The excretion rate was assumed to be 0.67 mgP m^{-3} d^{-1} at a fish population density of 10 g fresh weight m^{-3} and an individual weight of 200 g (Boers *et al.*, 1991). Kraft (1992) modelled the nutrient excretion of yellow perch (*Perca flavescens* (Mitchill)) in a system where this species was dominant and found the volumetric excretion of young-of-the-year fish to be far more important than older age classes. Because the fish composition changed towards younger individuals, we assumed a 50% higher excretion rate for 1991. The N excretion is calculated with the N : P ratio in the fish.

Boers *et al.* (1991) assumed a zero net growth for 1981 and 1987 and the total food uptake was set equal to excretion, using an assimilation efficiency of 75%. The other 25% was egested as faeces and became bottom detritus. Further, Boers *et al.* (1991) assumed that the fish diet consisted of 25% zooplankton and 75% zoobenthos. Gut analyses from 1989 indicated that zooplankton made up 12% of the diet of bream and 26% of the diet of roach. Roach is strongly dependent on zooplankton and was in poor condition as zooplankton was scarce. Based on the measured composition of fish species we assumed that only 16% of the diet consisted of zooplankton and that 84% of the diet was zoobenthos. After the food web manipulation the fish biomass increased with almost 70 kg ha^{-1} during the summer of 1991, resulting in a net uptake of nutrients. The total ingestion was calculated as the sum of growth and excretion divided by the assimilation efficiency. In this way 19% of the ingested food was used for growth. This is consistent with observations (E.H.R.R. Lammens, personal communication). Gut analysis during the summer of 1991 showed that 10% of the diet of bream and 60% of the diet of roach consisted of zooplankton. The composition of the fish stock changed. The amount of planktivorous fish was almost equal to the amount of benthic fish. Therefore, we estimated that 30% of the diet of fish was zooplankton and 70% was zoobenthos.

Many authors suggest that removal fish will decrease resuspension (e.g. Meijer *et al.*, 1994b). Boers *et al.* (1991) argued that in shallow lakes fish are not important in stirring sediment, compared with wind. This is confirmed by the observation that the inorganic matter content of the water did not significantly decrease in 1991 and 1992. Also Threlkeld (1994) argued that there is no sound experimental evidence that fish induce physical release of sediment bound P. Therefore, it cannot be expected that fish removal will affect the extinction in the water, nor significantly affect nutrient release from the sediments, via resuspension.

Zoobenthos

In 1991 the zoobenthos population was monitored. The dominant groups were Chironomidae, Oligochaeta and Gastropoda. Their occurrence was strongly dependent on the substratum. Chironomid biomass (July and September) on sandy, open spots was 0.21 g dry weight m^{-2}, while on macrophytes 0.39 g dry weight m^{-2} was found. A higher biomass is likely in the deeper, silty part of the lake. Chironomids consisted mainly of *Chironomous plumosus* (Linnaeus). The biomass was a little less than that found in 1989 and comparable with the biomass in a neighbouring lake without food web manipulation. Compared with 1989, the average biomass of the Oligochaeta increased from 1.9 to 4.3 g dry weight m^{-2} and gastropods increased from 0.03 to 0.3 g dry weight m^{-2}. The total zoobenthos biomass before and after food web manipulation was assumed to be 2.2 and 4.9 g dry weight m^{-2}. The P content was derived from Nakashima & Leggett (1980), and a N : P ratio of 10 by weight was assumed (Table 2).

We assumed that zoobenthos feeds on bottom detritus (not quantified), is eaten by fish (see 'Fish' above) and excretes to the dissolved nutrients pool in the water. Because biomass measurements agreed with previous estimations, we used the same excretion rates as Boers *et al.* (1991).

Macrophytes

From 1987 onwards the species composition and the percentage coverage of macrophytes were monitored and the biomass was estimated. The macrophytes consisted mainly of pond weeds (*Potamogeton pectinatus* L. and *P. perfoliatus* L.). The shoot biomass in 1987 was 8.3 g dry matter m^{-2}. This increased to 18.3 g dry matter m^{-2} in 1991. The root biomass was about the same as the shoot biomass. The growing season is from July to August. The nutrient contents of the dry matter were measured (Table 2). No data are available from 1981, but based on the low transparency of the water, it is assumed that there were no macrophytes.

In a review, Carpenter & Lodge (1986) concluded that macrophytes will predominantly act as a source for nutrients to the water column. The nutrients are taken up by the roots from the sediment interstitial water and released to the water after degeneration of the tissues. Direct release of organic P from living submersed macrophytes is ecologically insignificant. In a *P. pectinatus* stand, the P cycle is relatively closed (Howard-Williams & Allanson, 1981). P released by decaying macrophytes was rapidly assimilated by periphyton and epiphyton. During the growing season macrophytes contribute to the oxygenation of the sediments and may therefore retard the diffusion of P from the sediments to the overlying water.

Caffrey & Kemp (1992) demonstrated that the pond weed *P. perfoliatus* had a significant effect on N cycling by direct uptake of ammonium and nitrate and by indirect mechanisms leading to enhanced nitrification and denitrification. The proportion of nutrients taken up from pore water and overlying water is determined by the root : shoot biomass ratio and the ratio of the available nutrients in pore water and overlying water. In their experiments root uptake accounted for about 90% of the total N uptake in May and only 20% in July, although the specific uptake rate of roots was always higher than that of shoots. The average uptake for roots and shoots was 3.9 and 1.3 mgN d^{-1} g^{-1} dry weight, respectively. Further, most of the labelled N taken up by the roots was translocated to the shoots.

It is clear that macrophytes affected nutrient cycling directly by uptake and indirectly by influencing the conditions for several processes in the sediments. In the growing season they may be responsible for low dissolved concentrations, especially N, in the overlying water. Decay of the material will be responsible for a nutrient flux to the water. As no detailed data are available, uptake of nutrients from the water column by macrophytes was considered to equal the release of nutrients to the water after decay during the summer period. The indirect effects are incorporated in assumptions made by the release of nutrients from the sediments and the denitrification.

Benthic algae

Benthic algal chlorophyll was measured on the sediments in summer 1994 (Van Luijn *et al.*, 1995). The measured algal pigment composition differed from the water column, so the major part of the chlorophyll originated from benthic algae. From these data, we derived 100 mg m^{-2} as an average for 1991. Van Luijn *et al.* (1995) concluded that the growth of benthic algae was still limited by light availability. As almost no light reached the sediments in 1981, it is unlikely that benthic algae were present. Based on light penetration depth, we estimated that 50 mg chlorophyll m^{-2} was present in 1987 in benthic algae. To translate the data to nutrients, we used the same values as for non-cyanobacteria.

For systems with low dissolved nutrients in the water column and relatively enriched sediments, benthic algae are able to use nutrients from the sediments (Van Luijn *et al.*, 1995). Therefore, we assumed that benthic algae only interact with sediment nutrients, so their effect is incorporated in the measured internal loading.

Neomysis integer (Leach)
Although the freshwater shrimp *Neomysis integer* was present before the experiment, a significant biomass was found only in 1991. Increased numbers appeared in June and reached a maximum in August; the shrimps disappeared at the end of August. Their summer averaged numbers were around 100 ind. m^{-2}, corresponding to an estimated dry weight of 200 mg m^{-2}. This is almost 10-fold that in previous years. Meijer *et al.* (1994b) explained the low zooplankton biomass in summer as due to predation by *Neomysis*. Because no data were available on the nutrient content of *Neomysis*, we assumed that the ratios for zoobenthos will also hold for *Neomysis*. In 1992, analyses of gut contents indicated that both phytoplankton, zooplankton (mainly Copepoda and a few *Daphnia*) and macrofauna formed the diet of *Neomysis*. In 1991 *Neomysis* was abundant after the zooplankton peak. Further, high numbers of shrimps correlated with a high percentage of juveniles. Juvenile *Neomysis* feed on organic matter, so the impact on zooplankton predation in 1991 was possibly overestimated. Suppose the biomass *Neomysis* was reached within 1 month, the assimilation efficiency was 75% and the respiration was 10%, then, for a period of 30 days, a food uptake of 0.13 mgP m^{-3} d^{-1} can be calculated. This is small compared to other fluxes and, therefore, we did not take any *Neomysis* processes into account.

Other
No attention was paid to the role of birds and the mussel *Dreisena polymorpha* (Pallas). In 1989 - 1990 3.7 kg ha^{-1} of fish was consumed by cormorants (*Phalacrocorax carbo* (Linneaus)). This is only a small fraction of total biomass. In 1991 - 1992 the grazing increased to 12.5 kg ha^{-1} (Dirksen *et al.*, 1995), but this predation is already included in the food web manipulation (see 'Fish' above). Most feeding of waterbirds on macrophytes takes place in autumn and winter, when ducks and swans migrate from northern Europe. The effect of mussels on phytoplankton was considered small compared with grazing by zooplankton (R. Noordhuis, personal communication).

Results

The results of the food web analysis for P and N and for the different years are presented as summer averages in Tables 3 and 4. There is only a small difference between the uptake of dissolved nutrients (outflow, algal growth, and for N also denitrification) and the production of dissolved nutrients (respiration, excretion, mineralisation, autolysis and loadings). For P the differences between years were almost negligible. The removal of N was about 10% higher than the production in 1981.

Table 3 *Results of analyses of phosphorus (P) pools (italics) and fluxes in Wolderwijd, The Netherlands, for 1981, 1987 and 1991 and results for 1991 based on the study of Boers et al. (1991).*

	Units	1981	1987	1991	1991 (Boers)
Loadings					
External loading	$mgP\ m^{-3}\ d^{-1}$	1.6	1.1	0.5	0.4
Internal loading	$mgP\ m^{-3}\ d^{-1}$	3.5	1.1	0.3	1.5
Phytoplankton - biomass	$mgP\ m^{-3}$	*155*	*21.1*	*17.7*	*14*
Growth	$mgP\ m^{-3}\ d^{-1}$	27.8	11.4	11.5	12
Respiration	$mgP\ m^{-3}\ d^{-1}$	7.7	3.3	2.2	2.8
Sedimentation	$mgP\ m^{-3}\ d^{-1}$	4.6	0.6	0.5	
Mortality	$mgP\ m^{-3}\ d^{-1}$	13.1	2.4	6.2	0.9
Zooplankton - biomass	$mgP\ m^{-3}$	*3.1*	*8.1*	*8.6*	*14*
Grazing on phytoplankton	$mgP\ m^{-3}\ d^{-1}$	2.4	5.1	2.6	8.3
Detritus production	$mgP\ m^{-3}\ d^{-1}$	1.4	3.1	1.6	5.0
Excretion	$mgP\ m^{-3}\ d^{-1}$	0.4	1.1	1.1	1.8
Predation by fish	$mgP\ m^{-3}\ d^{-1}$	0.3	0.2	0.3	0.1
Growth	$mgP\ m^{-3}\ d^{-1}$	0.3	0.8	-0.3	1.4
Detritus					
Production rate	$mgP\ m^{-3}\ d^{-1}$	14.5	5.4	7.8	6.0
Autolysis	$mgP\ m^{-3}\ d^{-1}$	5.1	1.9	2.7	2.1
Production labile detritus	$mgP\ m^{-3}\ d^{-1}$	8.5	2.2	3.6	1.9
Production stabile detritus	$mgP\ m^{-3}\ d^{-1}$	0.9	1.3	1.5	2.0
Concentration labile detritus	$mgP\ m^{-3}$	*15.2*	*3.9*	*5.9*	*2.4*
Concentration stabile detritus	$mgP\ m^{-3}$	*15.7*	*22.4*	*14.7*	*6.7*
Mineralization labile detritus	$mgP\ m^{-3}\ d^{-1}$	7.6	2.0	3.0	1.2
Sedimentation labile detritus	$mgP\ m^{-3}\ d^{-1}$	0.9	0.2	0.6	0.7
Sedimentation stabile detritus	$mgP\ m^{-3}\ d^{-1}$	0.9	1.3	1.5	2.0
Fish - biomass	$mgP\ m^{-3}$	*131*	*89*	*35*	*20*
Net growth	$mgP\ m^{-3}\ d^{-1}$	0.0	0.0	0.2	
Excretion	$mgP\ m^{-3}\ d^{-1}$	1.3	0.9	0.5	0.2
Egestion	$mgP\ m^{-3}\ d^{-1}$	0.4	0.3	0.2	0.1
Zoobenthos - biomass	$mgP\ m^{-3}$	*14.3*	*14.3*	*31.8*	*30*
Predation by fish	$mgP\ m^{-3}$	1.5	1.0	0.6	0.2
Excretion	$mgP\ m^{-3}\ d^{-1}$	0.7	0.7	1.3	1.3
Macrophytes - biomass	$mgP\ m^{-3}$	*0*	*12*	*26*	
Benthic algae - biomass	$mgP\ m^{-3}$	*0*	*21*	*39*	
Neomysis integer - biomass	$mgP\ m^{-3}$	*0.1*	*0.1*	*1.3*	

Table 4 *Results of analyses of nitrogen (N) pools (italics) and fluxes in Wolderwijd, The Netherlands, for 1981, 1987 and 1991.*

	Units	1981	1987	1991
Loadings				
External loading	mgN m^{-3} d^{-1}	17.7	20.2	15.9
Internal loading	mgN m^{-3} d^{-1}	13.7	4.1	2.3
Denitrification	mgN m^{-3} d^{-1}			15.0
Phytoplankton - biomass	mgN m^{-3}	*1856*	*253*	*213*
Growth	mgN m^{-3} d^{-1}	334	137	138
Respiration	mgN m^{-3} d^{-1}	92.8	39.2	25.9
Sedimentation	mgN m^{-3} d^{-1}	55.7	7.6	6.4
Mortality	mgN m^{-3} d^{-1}	157	28.5	74.0
Zooplankton - biomass	mgN m^{-3}	*28.9*	*75.6*	*80.0*
Grazing on phytoplankton	mgN m^{-3} d^{-1}	28.6	61.2	31.7
Detritus production	mgN m^{-3} d^{-1}	17.2	36.7	19.0
Excretion	mgN m^{-3} d^{-1}	3.8	9.8	10.4
Predation by fish	mgN m^{-3} d^{-1}	2.8	1.9	2.4
Growth	mgN m^{-3} d^{-1}	4.9	12.8	0.0
Detritus				
Production rate	mgN m^{-3} d^{-1}	174	65.2	93.0
Autolysis	mgN m^{-3} d^{-1}	61.0	22.8	32.6
Production labile detritus	mgN m^{-3} d^{-1}	102	26.3	42.8
Production stabile detritus	mgN m^{-3} d^{-1}	11.3	16.1	17.7
Concentration labile detritus	mgN m^{-3}	*182*	*46.9*	*71.4*
Concentration stabile detritus	mgN m^{-3}	*189*	*269*	*177*
Mineralization labile detritus	mgN m^{-3} d^{-1}	91.0	23.5	35.7
Sedimentation labile detritus	mgN m^{-3} d^{-1}	10.9	2.8	7.1
Sedimentation stabile detritus	mgN m^{-3} d^{-1}	11.3	16.1	17.7
Fish - biomass	mgN m^{-3}	*516*	*349*	*138*
Net growth	mgN m^{-3} d^{-1}	0.0	0.0	0.7
Excretion	mgN m^{-3} d^{-1}	13.1	8.8	5.2
Egestion	mgN m^{-3} d^{-1}	4.4	2.9	2.0
Zoobenthos - biomass	mgN m^{-3}	*143*	*143*	*318*
Predation by fish	mgN m^{-3}	14.6	9.9	5.5
Excretion	mgN m^{-3} d^{-1}	6.7	6.7	13.4
Macrophytes - biomass	mgN m^{-3}	*0.0*	*94*	*208*
Benthic algae - biomass	mgN m^{-3}	*0.0*	*257*	*471*
Neomysis integer - biomass	mgN m^{-3}	*1.3*	*1.3*	*13.0*

For all years studied, it was implicitly assumed that biomasses and fluxes were constant over the summer. As can be seen in Figure 4 for the zooplankton biomass and chlorophyll, this was not the case. Macrophytes have a distinctive growing and decaying period within the summer. Furthermore, fluxes are not proportional to biomass. In general, rates tend to decrease at high biomass due to saturation effects and food limitations. Compared with the uncertainty in some of the coefficients, however, the accuracy of the results of the nutrient cycle analyses is acceptable.

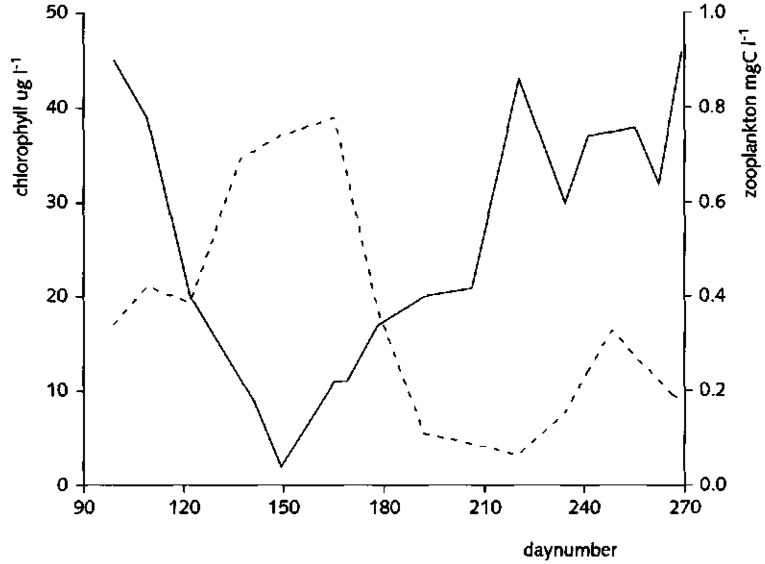


Figure 4 *Chlorophyll (solid line) and zooplankton (dashed line) biomass of Wolderwijd in the summer of 1991.*

There were minor differences in the biomass of the variables and in the fluxes for 1981 and 1987 between this study and the results of Boers *et al.* (1991)(Table 3). Some calculation errors in the previous study have been corrected and the averaging procedure of variables measured in both parts of the lake (Wolderwijd and Nuldernauw) has been improved. The C : P ratio of phytoplankton was adjusted and the sedimentation of phytoplankton was taken into account.

Some minor deviations between this study and the study of Boers *et al.* (1991) for the results for 1991 can be explained in a similar way. However, the most important differences between this and the previous study are caused by a different development of the lake ecosystem than that forecasted by Boers *et al.* (1991), as revealed by the intensive monitoring programme in 1991. Except for the dissolved nutrients and the nutrients in zooplankton and zoobenthos, all pools were underestimated by Boers *et al.* (1991) (Figure 5). The zooplankton biomass was significantly lower than in the previous study, but matches with observations in other biomanipulated lakes (Gulati, 1990). The estimation of the development of the zoobenthos is in agreement with observations. In 1991 benthic algae formed more than 50% of the total nutrients in the benthos and were not included in the study of Boers *et al.* (1991). The composition of phytoplankton and fish differed from that predicted. In

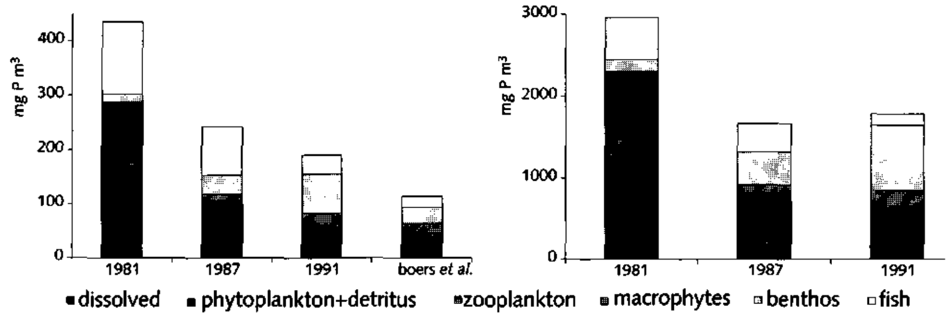


Figure 5 *Phosphorus (left) and nitrogen (right) pools in Wolderwijd, averaged over the summer for years 1981, 1987 and 1991; for phosphorus in 1991 the results of Boers et al. (1991) are also depicted.*

1991 more than 50% of the biovolume was cyanobacteria, while 0% had been anticipated. The percentage of planktivorous fish was 55% instead of 25%. The amount of fish just after the removal corresponded to the target and the estimation of Boers *et al.* (1991). However, during the summer biomass increased rapidly.

Further, most of the fluxes were different from the previous study. The internal P loading from the sediments is low compared to the assumption made by Boers *et al.* (1991). This may be explained by the increase of benthic algae and macrophytes. Benthic algae are capable of reducing nutrient release from sediments by direct and indirect means (Van Luijn *et al.*, 1995). The low grazing activity of zooplankton in 1991 resulted in a negative growth. N fluxes were strongly coupled to the P fluxes, because most assumptions were based on a constant N : P ratio per variable.

Discussion

It is well known that it takes several years for a lake to reach a new equilibrium after a decrease of the external P loading or food web manipulation (Jeppesen *et al.*, 1990; Carpenter *et al.*, 1992; Van der Molen & Boers, 1994). Water quality data from 1992 and 1993 show that these years were comparable to 1991 with respect to the nutrient and chlorophyll concentration and the algal species composition. The clear water period of 6 weeks, which was observed in early summer 1991, decreased to 2 weeks in 1992 and was absent in the next two years. Also, developments in the structure of the ecosystem indicate that the lake was not yet in a steady state. The density of the macrophyte coverage increased after 1991 and also the fish population increased again, in spite of additional fishing. With respect to low phytoplankton biomass, low fish biomass and increased transparency, however, 1991 can be considered as the optimal result after the food web manipulation in Wolderwijd.

From 1981 to 1987 the lake changed from a hypertrophic, cyanobacteria-dominated system towards a eutrophic lake with a succession of algal species. The decrease of phytoplankton was mainly responsible for the decrease in the total nutrient pool of the system. The nutrient flow through the food web is dominated by algal uptake of dissolved nutrients, transformation of phytoplankton to detritus and autolysis and mineralisation to the dissolved pool of nutrients (Figure 6). The decreases in dissolved nutrients and nutrients in fish were responsible for a decrease in the total P pool from 1987 to 1991. This was compensated by an increase in nutrients in the benthos and in macrophytes, which caused the total N pool of 1991 to be even higher than in 1987. The main nutrient fluxes through the food web decreased in 1991 (Figure 7) compared to 1981, but the dominating pathways were still the same as in 1981.

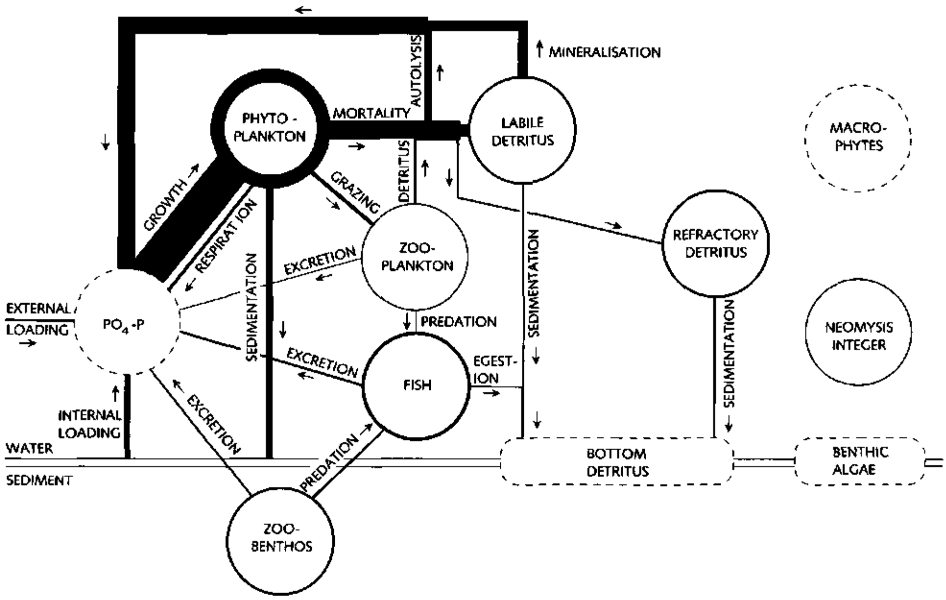


Figure 6 *Phosphorus pools and fluxes for Wolderwijd in 1981. The thickness of the lines illustrate the relative size of the pools and fluxes from Table 3, but for the conversion of pools and fluxes a different factor is used.*

Fish, zoobenthos, macrophytes and benthic algae were, apart from the sediments, the most important pools of P for Wolderwijd in 1991. However, the fluxes were low compared with the size of these pools. The excretion of fish is of the same order of magnitude as the external loading, but low compared with the cycling of nutrients by respiration of algae and autolysis and mineralisation of detritus. The removal of fish resulted in a significant decrease in the total nutrient pool in the water, but this was not sufficient to eliminate phytoplankton via nutrient stress. Based on data of Caffrey & Kemp (1992) and measurements, macrophytes can take up about 15 mgN m^{-3} d^{-1} during the growing season. In parts of the lake with a high coverage of macrophytes this may be more. Macrophytes are able to consume the available N from the water column. In Wolderwijd bioassay experiments for phytoplankton growth indeed indicated N limitation. However, for the total lake and the whole summer period the assumption of equilibrium between uptake and release of N seems to be valid. Further, we have no indications yet that the development of macrophytes influenced P cycling in Wolderwijd to any extent. It is obvious that different macrophyte species have a different impact on nutrient cycling. In Wolderwijd *Potamogeton* sp. has been progressively replaced by Characeae since 1991. The change to a stable community of submerged macrophytes will probably take many years.

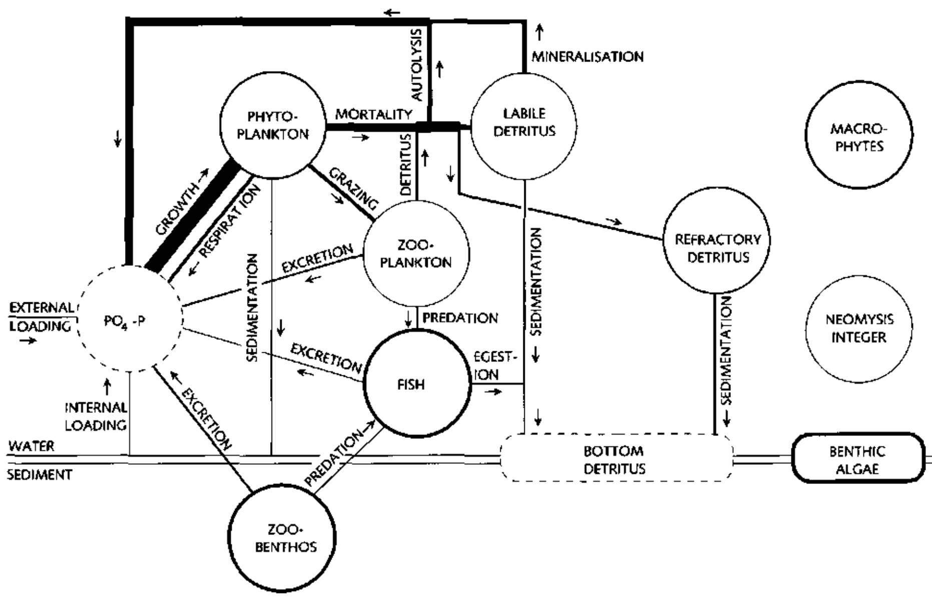


Figure 7 *Phosphorus pools and fluxes for Wolderwijd in 1991. The thickness of the lines illustrate the relative size of the pools and fluxes from Table 3, but for the conversion of pools and fluxes a different factor is used.*

The production : biomass ratio for phytoplankton increased in time, while the grazing : biomass ratio of zooplankton decreased (Figure 8). The average zooplankton biomass did not increase significantly from 1987 to 1991 and the growth of zooplankton was almost zero in 1991. The measured average ingestion rate was less than assumed by Boers *et al.* (1991). Availability of food (phytoplankton) was higher in 1991, except for a few weeks with an almost complete absence of phytoplankton. The most likely explanation of the low efficiency of zooplankton was the relative high percentage of cyanobacteria. This food does not favour zooplankton (Gulati, 1990), and the grazing pressure on zooplankton results in their disappearance.

In the model, the N : P ratio for zooplankton is lower than that for phytoplankton. To adjust this, zooplankton will excrete more N relative to P. This may be an important competitive factor determining the nutrient limitation of algae and therefore influencing the algal species composition (Carpenter *et al.*, 1992; Sterner *et al.*, 1992). The excretion fluxes are small or equal compared with independent fluxes as the loadings, so variations in the ratio of nutrient excretion is not expected to be important for Wolderwijd.

The results of the model demonstrate clearly why the Secchi depth increased so little since 1981. Due to the change in algal species from 100% cyanobacteria in 1981 to 52% cyanobacteria in 1991, relatively more stable detritus was produced. The concentration of chlorophyll decreased from 1981 to 1991 by a factor of almost nine, while the concentration detritus decreased only by a third. The concentration inorganic matter was unaffected by the food web manipulation. From 1987 onwards detritus and inorganic matter determined Secchi disc transparency and the contribution of phytoplankton was only small.

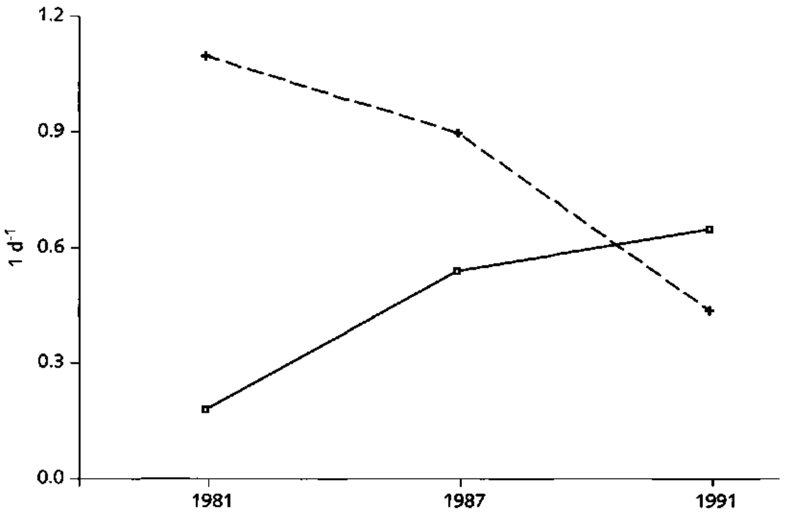


Figure 8 *Production : biomass ratio for phytoplankton (solid line) and grazing : biomass ratio of zooplankton (dashed line) in Wolderwijd for 1981, 1987 and 1991.*

Jeppesen *et al.* (1990) empirically derived a threshold P concentration of 0.08 - 0.15 mg l^{-1}, below which food web manipulation in lakes > 10 ha may have an effect in the long term. Klinge *et al.* (1994) derived a concentration of 0.1 mg l^{-1}, below which predator consumption should be able to exceed the prey production in large lakes. At this level, however, food web manipulation was not successful in Wolderwijd. The water of Wolderwijd became clear for 6 weeks after the food web manipulation experiment due to grazing by zooplankton, but this effect did not last over the summer and was almost absent in the following years. The short period of clear water initiated the growth of macrophytes. Because predators did not develop, no structural change in fish species composition was achieved. The differences between the nutrient cycle of 1987 and 1991 were mainly caused by a gradual change driven by low P input.

References

Boers, P.C.M., L. Van Ballegooijen & E.J.B. Uunk, 1991. Changes in phosphorus cycling in a shallow lake due to food web manipulations. *Freshwater Biology* 25: 9-20.

Boers, P.C.M. & D.T. Van der Molen, 1993. Control of eutrophication in lakes: the state of the art in Europe. *European Water Pollution Control* 2: 19-25.

Boers, P.C.M. & O. Van Hese, 1988. Phosphorus release from the peaty sediments of the Loosdrecht Lakes (The Netherlands). *Water Research* 22: 355-363.

Caffrey, J.M. & M. Kemp, 1992. Influence of the submersed plant, *Potamogeton perfoliatus*, on nitrogen cycling in estuarine sediments. *Limnology and Oceanography* 37: 1483-1495.

Carpenter, S.R., K.L. Cottingham & D.E. Schindler, 1992. Biotic feedbacks in lake phosphorus cycles. *TRends in Ecology and Evolution* 7: 332-336.

Carpenter, S.R. & D.M. Lodge, 1986. Effects of submersed macrophytes on ecosystem processes. *Aquatic Botany* 26: 341-370.

Dirksen, S., T.J. Boudewijn, R. Noordhuis & E.C.L. Marteijn, 1995. Cormorant *Phalacrorax carbo sinensis* in shallow eutrophic freshwater lakes: prey choice and fish consumption in the non-breeding period and effects of large-scale fish removal. *ARDEA Journal of The Netherlands Ornithologists' Union* 83: 167-184.

Gulati, R.D., K. Siewertsen & G. Postema, 1982. The zooplankton: its community structure, food and feeding, and role in the ecosystem of Lake Vechten. *Hydrobiologia* 95: 127-163.

Gulati, R.D., 1990. Structural and grazing responses of zooplankton community to biomanipulation of some Dutch water bodies. *Hydrobiologia* 200/201: 99-118.

Gunnison, D. & M. Alexander, 1975. Resistance and susceptibility of algae to decomposition by natural microbial communities. *Limnology and Oceanography* 20: 64-70.

Howard-Williams, C. & B.R. Allanson, 1981. Phosphorus cycling in a dense *Potamogeton pectinatus*. L. bed. *Oecologia* 49: 56-66.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer & K. Olrik, 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic temperate lakes 2: threshold levels, long-term stability and conclusions. *Hydrobiologia* 200/201: 219-227.

Jewell, W.J. & P.L. McCarthy, 1971. Aerobic decomposition of algae. *Environmental Science and Technology* 5: 1023-1031.

Klinge, M., M.P. Grimm & S.H. Hosper, 1994. Eutrophication and ecological rehabilitation of Dutch lakes: explanation and prediction by a new conceptual framework. In: Living with Water. International Conference on Integrated Water Resources Management, Sep. 26-29 1994, Amsterdam, The Netherlands. pp. 313-325.

Kraft, C.E., 1992. Estimates of phosphorus and nitrogen cycling by fish using a bioenergetics approach. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 2596-2604.

Los, F.J., 1991. Mathematical simulation of algae blooms by the model BLOOM II. T68, Delft Hydraulics, The Netherlands. 113 pp.

Meijer, M.-L, E. Jeppesen, E. Van Donk, B. Moss, M. Scheffer, E.H.R.R. Lammens, E.H. Van Nes, J.A. Van Berkum, G.L. De Jong, B.A. Faafeng & J.P. Jensen, 1994a. Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five-year results of four biomanipulation cases in The Netherlands and Denmark. *Hydrobiologia* 275/276: 457-466.

Meijer, M.-L, E.H. Van Nes, E.H.R.R. Lammens, R.D. Gulati, M.P. Grimm, J. Backx, P. Hollebeek, E.M. Blaauw & A.W. Breukelaar, 1994b. The consequences of a drastic fish stock reduction in the large and shallow Lake Wolderwijd, The Netherlands. Can we understand what happened? *Hydrobiologia* 275/276: 31-42.

Nakashima, B.S. & W.C. Leggett, 1980. The role of fish in the regulation of phosphorus availability in lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 1540-1549.

Penczak, T. & I. Tatrai, 1985. Contribution of bream (*Abramis brama* L.) to the nutrient dynamics of Lake Balaton. *Hydrobiologia*, 126, 59-64.

Scavia, D., 1980. An ecological model of lake Ontario. *Ecological Modelling* 8: 49-78.

Sterner, W.S., J.J. Elser & D.O. Hessen, 1992. Stoichiometric relationships among producers, consumers and nutrient cycling in pelagic ecosystems. *Biochemistry* 17: 49-67.

Threlkeld, S.T., 1994. Benthic-pelagic interactions in shallow water columns: an experimentalist's perspective. *Hydrobiologia* 275/276: 293-300.

Van der Molen, D.T. & P.C.M. Boers, 1994. Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. *Hydrobiologia* 275/276: 379-389.

Van Luijn, F., D.T. Van der Molen, W. Luttmer & P.C.M. Boers, 1995. The influence of benthic diatoms on the nutrient release from sediments of shallow lakes recovering from eutrophication. *Environmental Science and Technology* 32: 89-97.

Zevenboom, W., 1980. Growth and nutrient uptake kinetics of Oscillatoria agardhii. Thesis University of Amsterdam, The Netherlands.

Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, The Netherlands

D.T. Van der Molen[1], R. Portielje[1], P.C.M. Boers[1] & L. Lijklema[2], 1998

Water Research 32: 3281-3288.

[1] Institute for Inland Water Management and Waste Water Treatment, Lelystad, The Netherlands
[2] Ketsheuvel 33, 6871 EB Renkum, The Netherlands

Abstract

Since the creation of Lake Veluwe, The Netherlands, in 1956, a period of eutrophication has been followed by a period of oligotrophication. Around 1970, the lake switched from clear to turbid and recently from turbid to clear. The effect of eutrophication and oligotrophication has been studied with a conceptual model, based on the mass balance of total phosphorus in the top layer of the sediment. The model has been applied to the shallow part and the deeper part of Lake Veluwe. Whereas drastic changes in the external phosphorus loading, and concomitantly the phosphorus concentration in the lake water, occurred, the phosphorus pool in the sediment top layer only responds gradually to changed boundary conditions. The phosphorus content in the sediment top layer of the deeper part of Lake Veluwe doubled between 1956 and 1979. After 1979 it decreased. Losses were due to infiltration of dissolved phosphorus and burial of solid phosphorus caused by net sedimentation with material from the shallow part of the lake. Sensitivity analyses revealed that sediment phosphorus is also primary affected by these two factors. Based on the model predictions it is expected that in 2006 the phosphorus content of the sediment will be below the phosphorus content at creation of Lake Veluwe. Changes in flushing of the lake will probably have little effect on sediment phosphorus, while development of macrophytes will cause higher phosphorus contents in the shallow areas. However, with the current boundary conditions further ecological improvement of Lake Veluwe may be expected. The phosphorus pool in the sediment top layer is an important determinant in the long-term development of shallow lake ecosystems and, therefore, the modelling concept may also be applicable to other shallow lakes.

Introduction

Eutrophication, or nutrient enrichment, of freshwater lakes eventually results in turbid, phytoplankton dominated systems. Analogously, oligotrophication can be defined as nutrient amelioration. In lake management practice, oligotrophication can be achieved by a decreased external nutrient loading. In shallow lakes, this will ultimately result in clear water and primary production dominated by submerged macrophytes. Both the turbid and clear water state are stable in the sense that several buffer mechanisms protect the system against perturbations (Scheffer, 1989). A hysteresis effect can be observed when a shallow lake is first enriched with nutrients, followed by reduction of the nutrient loading (Figure 1). To promote the re-establishment of the desired clear water phase, several 'system disturbances' can be envisaged (B; Figure 1), such as food web manipulation, chemical manipulation and dredging (Benndorf, 1987; Cooke *et al.*, 1993; Boers *et al.*, 1994). These system disturbances are mainly successful in small water bodies. In shallow lakes larger than a few square kilometres the internal phosphorus loading and resuspension of sediment are strong stabilising factors supporting the maintenance of eutrophic, turbid conditions. Here, it may take several years before a new steady state is reached following a changed external loading (Jeppesen *et al.*, 1990, 1991; Meijer *et al.*, 1994; Van der Molen & Boers, 1994).

Therefore, information on the nutrient status, mainly phosphorus (P), is essential to predict the development of a lake under present or future P loading and to forecast the effect of additional

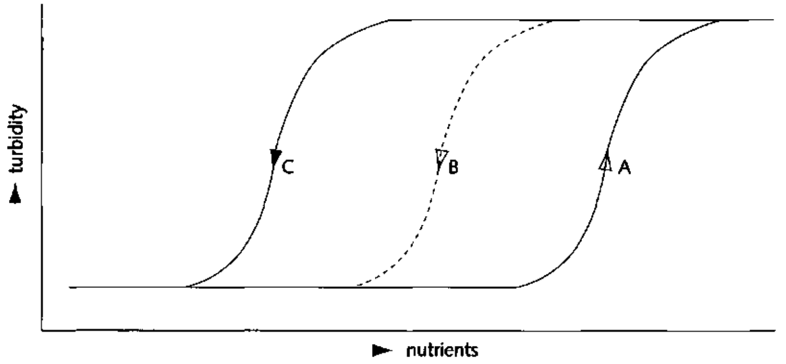


Figure 1 *Hysteresis effect during recovery from eutrophication (Scheffer, 1989); response of lake turbidity on an increase of nutrients (A), a decrease of nutrients (C) and a shortcut by 'system disturbances' (B).*

restoration measures. This nutrient status consists of the nutrients available in the water and the top layer of the sediment. The amount of P in the overlying water is generally small compared to the amount of P in the top layer of sediment.

In this study the conceptual mass balance model of Lijklema (1982) has been extended and applied to study the amount of P in the sediment top layer of Lake Veluwe, The Netherlands, in relation to several external and internal conditions. The top layer of the sediment interacts with the overlying water and the availability of total P in the lake increases if conditions are favourable for desorption of phosphate (Boström *et al.*, 1988). Therefore, total P in the sediment top layer has been taken as the main variable in the conceptual model. Lake Veluwe is a well-documented example of a large, shallow lake where a period of eutrophication has been succeeded by almost two decades of oligotrophication. The lake followed line A in Figure 1 from left to right from the late sixties to 1979. Since then it followed a long way of oligotrophication and presently the state of Lake Veluwe is expected to be somewhere in the descent of line C.

The sensitivity of the sediment P content for boundary conditions and model parameters was analysed. Finally, the model was used to speculate about the effects of flushing and the recovery of macrophytes on sediment P.

Case study Lake Veluwe

History

Lake Veluwe was created in 1956 by the formation of polders. Together with adjacent 'border lakes', it prevents unacceptable groundwater losses from the 'old' land. The lake has a unique hydrology due to the location between the 'old' land and the polder, lying about 5 m below mean sea level. The part of the lake bordering the 'old' land is shallow (< 1.0 m), with a sandy sediment and an inflow of groundwater and 19 streams. The rest of the lake is deeper (1 - 3 m, with some pits down to 5 m depth), has a more silty sediment and water infiltrates to the polder. The lake has a total surface area of 32.8 km^2.

In its first years after creation the water was clear and there was a rich animal and plant life (Noordhuis, 1997). Macrophytes were dominated by *Characeae*. In the second half of the sixties the water quality deteriorated due to an increased nutrient loading. The water became turbid with Secchi depth transparency less than 0.2 m. In the early seventies the algal species composition was dominated by cyanobacteria, especially *Oscillatoria agardhii* (Berger & Bij de Vaate, 1983). Submerged macrophytes had by that time disappeared. A monitoring and restoration program started in the second half of the seventies (Hosper, 1984; Jagtman *et al.*, 1992; Noordhuis, 1997). In 1979 P elimination was introduced at the sewage treatment plant discharging into Lake Veluwe. Also since 1979 the lake has been flushed in winter with polder water. This water is poor in P and algae. The P and chlorophyll-a concentrations responded within a year, but the transparency remained low. Since the beginning of the nineties ecological recovery has been observed (Noordhuis, 1997). Macrophytes reappeared, macrofauna and fish densities and compositions altered and the water is clear in a large part of the lake (Figure 2).

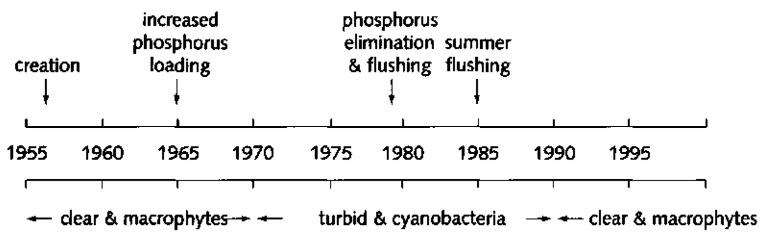


Figure 2 *The history of Lake Veluwe, The Netherlands.*

Conceptual model

Several deterministic models, describing sediment P, have been developed in the past decades (e.g. Berner, 1974; Jørgensen *et al.*, 1982; Kamp-Nielsen, 1983; Smits & Van der Molen, 1993). In the present study, boundary conditions for the development of the sediment P model were closed mass balances and a complexity that is justified by data availability. Further, it was assumed a priori that seepage and infiltration could be important for sediment P in Lake Veluwe. Interactions of sediment phosphate with the overlying water and temporal variations within a year were not objectives for this study. The model is derived from the mass balance of P over the sediment top layer as described by Lijklema (1982):

$$\frac{dc}{dt} = \frac{M}{L} - c \cdot \frac{\Delta L}{L} \qquad (1)$$

The total P concentration in the sediment top layer, c, is expressed in gP m^{-3}. L and ΔL are the mixing depth (m) and the burial rate (m y^{-1}). M is the net P accumulation rate (gP m^{-2} y^{-1}) in the top layer of the sediment. The accumulation rate is related to the net annual P retention (R), defined as the difference between P entering and leaving the system with surface flow. The net retention includes sedimentation and resuspension of particulate P and diffusive and advective transport of dissolved P over the sediment - water interface. Seepage (S) and infiltration (I) not only occur at the sediment - water interface (z=0), but also at the imaginary interface between the top layer of the sediment and the deeper sediment (z=Z). Therefore, the net P accumulation in the top layer of the sediment is:

$$M = R + S_z - I_z \tag{2}$$

In Lake Veluwe seepage and infiltration occur simultaneously in spatial separated areas (Figure 3). Seepage takes place in the shallow part of the lake, denoted by 's', and infiltration in the deeper part of the lake, indicated by 'd'. Both parts cover about 50% of the lake (PER, 1986). Equation 1 is applied to both parts of the lakes separately.

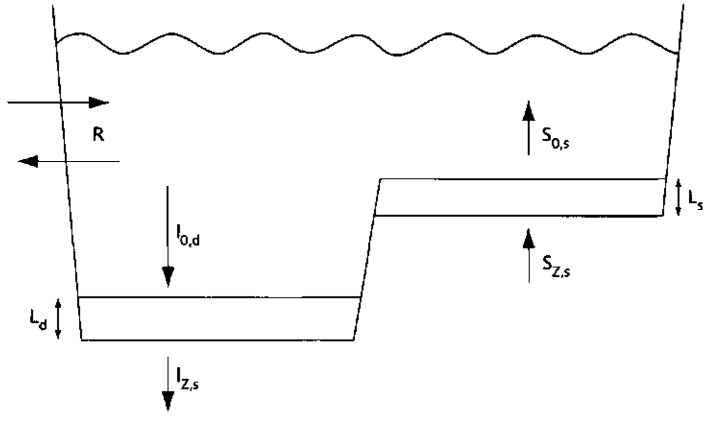


Figure 3 *Major P fluxes in a mass balance model of Lake Veluwe.*

The sediment in the shallow part of the lake consists mainly of clean sand. Therefore, a net resuspension (erosion) is assumed for this part of the lake. Supply of P to the sediment top layer is caused by seepage and by the 'addition' of sediment from the deeper layer to the top layer due to erosion. Equation 1 becomes:

$$\frac{dc_s}{dt} = \frac{(S_{Z,s} - S_{0,s})}{L_s} - (c_{0,s} - c_{Z,s}) \cdot \frac{\Delta L_s}{L_s} \tag{3}$$

Net sedimentation of solids causes burial of P in the sediment. Burial results in an upward shift of the sediment - water interface and in a removal of solid P at the interface between the sediment top layer and the deeper sediment. It is assumed that net sedimentation of particulate matter only occurs in the deeper part of the lake. Here, M is determined by the P retention in the system, transport of material from the shallow part of the lake (redistribution) and loss of P by infiltration.

The P accumulation by redistribution equals the erosion of P from the shallow part of the lake ($E_s = \Delta L_s \cdot c_s$; E in gP m^{-2} y^{-1}). The sediment P content in the deeper part of the lake can now be described by combination of equation 1 and 2:

$$\frac{dc_d}{dt} = \frac{(R + E_s - I_{Z,d})}{L_s} - c_d \cdot \frac{\Delta L_d}{L_s} \qquad (4)$$

Phosphorus loading and retention

Lake Veluwe has been intensively monitored. Water balance data are available from 1957 onwards (Noordhuis, 1997). More detailed data on the water balance and the P mass balance are available since 1976 (PER, 1986; Hosper, 1984; Jagtman *et al.*, 1992; Van Ballegooijen & Van der Molen, 1994). The external P loading before 1976 is estimated from data over the period 1976 - 1978 and unpublished data. Figure 4 depicts the surface external P loading and the P retention from 1956 - 1996. The P retention before 1976 is estimated at 80% of the surface external loading. This percentage is the average of the retention calculated for 1976 - 1979 and 1983. These years had similar, low flushing rates compared to the period before 1976. The high retention in 1976 - 1978 was caused by low surface outflow. Infiltration and evaporation compensated for the inflow of water to the lake. P elimination at the sewage treatment plant decreased the external P loading from 3 to 1 gP m^{-2} y^{-1} in 1979. Flushing with polder water in the winter from 1979 onwards caused an increase in the P loading from this source, but the average hydraulic residence time decreased from 0.45 to 0.20 y. After 1979, the concentration increased during summer due to a relatively high internal loading of phosphate from the sediment and due to concentration of dissolved components caused by evaporation. Additional flushing in the summer from 1985 onwards resulted in a further decrease of the hydraulic retention time to 0.15 y. Mass balance data from after 1992 were not available. There were no significant changes in the external conditions, so from 1993 onwards the data from 1992 were used.

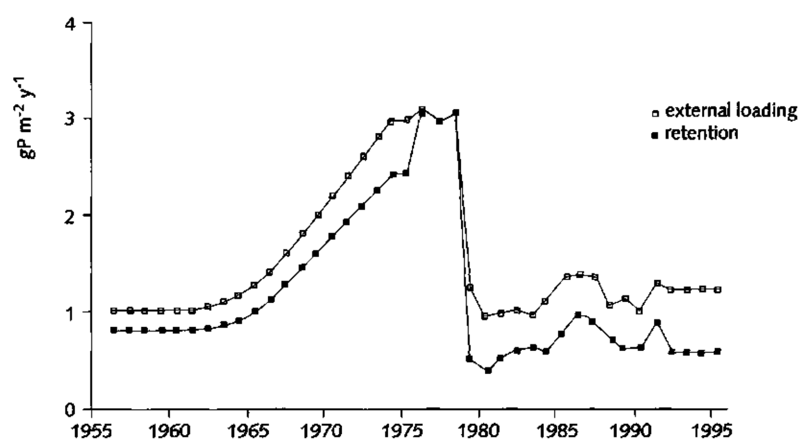


Figure 4 *Yearly averaged external P loading and retention of Lake Veluwe since the creation of the lake.*

Seepage and infiltration

Hydraulic loading by seepage and infiltration was calculated with a groundwater model (De Lange, 1996). Seepage in the shallow part of the lake is estimated at $27.2 \cdot 10^6$ m^3 y^{-1} and infiltration in the deeper part of the lake at $90 \cdot 10^6$ m^3 y^{-1}. The P concentration in the seepage water was measured in 1990 at a depth of 1 meter below the sediment surface (0.08 gP m^{-3}, standard deviation 0.07; Van Ballegooijen & Van der Molen, 1994), and is assumed to be constant over the whole period. This results in a flux of 0.134 gP m^{-2} y^{-1}, averaged over the shallow part of the lake. P lost by infiltration is determined by the dissolved P concentration at the interface between the top layer and the deeper sediment. It is assumed that this concentration is related to sediment P according to the Langmuir equation. The three parameters of this relation are the maximum adsorption capacity of the sediment, the Langmuir adsorption constant and the ratio between adsorbed and total P. Values of 0.1 gP kg^{-1} dry weight, 2.8 m^3 g^{-1}P and 0.1 (-) are used, respectively, derived from measurements in sediment of Lake Veluwe (Brinkman & Van Raaphorst, 1986, p. 257, 272). Danen-Louwerse *et al.* (1993) reported similar values. As a result, the estimated porewater P concentration is in agreement with measurements (Brinkman & Van Raaphorst, 1986; Danen-Louwerse *et al.*, 1996).

Burial

Several processes determine the net sedimentation or burial: external supply of suspended matter, surface outflow of solids and formation and dissolution of solids in the water. Redistribution of solids by erosion and sedimentation causes variation of net sedimentation within the lake.

External input of solids consists of surface water loading and atmospheric deposition. It is assumed that in Lake Veluwe the latter is negligible compared to the external input. The two dominating external sources of suspended matter are streams and flushing. The loading of inorganic matter from streams was estimated from data of 1984 - 1986. The loading of solids imported by flushing was derived from discharges over the period 1976 - 1992 and the suspended solids concentration measured in 1982 - 1986 (Van der Molen & Boers, 1992). The annual import of solids increased after 1979 due to additional flushing. Based on import of dissolved substances and sediment characteristics, it is assumed that calcareous compounds dominated the formation of solids in Lake Veluwe (Brinkman & Van Raaphorst, 1986). Mass balance data of calcium are available for 1981 and 1985 - 1987. In 1981 the surface inflow and outflow of calcium were $27.0 \cdot 10^6$ and $29.8 \cdot 10^6$ kg dry weight respectively (Van der Molen & Boers, 1992). This indicates a net dissolution of calcite from the sediment of $7.0 \cdot 10^6$ kg dry weight. Analogously, net calcite precipitation was estimated for 1985, 1986 and 1987 at 23.4, 1.0 and $4.6 \cdot 10^6$ kg dry weight, respectively (Danen-Louwerse *et al.*, 1995). The average net calcium precipitation based on data of these four years is used to estimate calcite sedimentation after 1979. Because most of the calcium is imported with flushing water, it is assumed there was no net formation of calcite before 1979. The outflow of solids from the lake is estimated from data of 1981 and 1978 (Table 1). Net accumulation of solids resulted in sediment growth in the deeper part of the lake of $0.01 \cdot 10^{-3}$ m y^{-1} before 1980 and $0.39 \cdot 10^{-3}$ m y^{-1} after 1980, assuming a sediment density 700 kg dry weight m^{-3} (Brinkman & Van Raaphorst, 1986).

Table 1 *Solids mass balance for Lake Veluwe*

10^6 kg y^{-1}	< 1979	> 1979
External loading -streams	0.40	0.40
External loading - flushing	0.13	0.53
Net sedimentation of calcite	0.0	5.5
Outflow	-0.40	-2.0
Total	0.13	4.43

Besides sediment growth, there is a redistribution of solids due to erosion and sedimentation. In 1981, it was estimated that about $1.0 \cdot 10^6$ m^3 solids were added to deeper parts of the lake since its creation in 1956 (Van der Molen & Boers, 1992). This results in a sediment growth (ΔL_d) of $2.6 \cdot 10^{-3}$ m y^{-1} in the deeper part of the lake. Because of the higher sediment density in the shallow part of the lake (1400 kg dry weight m^{-3}; Brinkman & Van Raaphorst, 1986), the erosion rate in the shallow part amounts $1.3 \cdot 10^{-3}$ m y^{-1} (ΔL_s). In conclusion, import and formation of solids are small compared to the internal redistribution of solids. Therefore only this internal redistribution is taken into account for the burial rate in the model.

Mixing depth and initial conditions

The total P content observed in slices of 0.02 m of the upper 0.14 m of the sediment was almost constant (Brinkman & Van Raaphorst, 1986; p. 248). Therefore, the sediment top layer is assumed to be completely mixed for total P. In porewater concentration profiles generally no gradients were observed for dissolved P and nitrogen below a depth of 0.088 m (Danen-Louwerse *et al.*, 1996). Therefore this depth will be used as the mixing depth in the model. The initial total P content (in 1956) in the deeper part of the lake is estimated from the lowest values reported by Brinkman & Van Raaphorst (1986) at 0.3 gP kg^{-1} dry weight.

Results

In the shallow part of Lake Veluwe the P content in the sediment was calculated with equation 3. Because in the mixing layer c is constant, $c_{Z,s} = c_{0,s}$, and there will be equilibrium if $S_{Z,s} = S_{0,s}$. Smits & Van der Molen (1993) reported an observed P content of about 0.08 gP kg^{-1} dry weight, based on several measurements in 1979 - 1980, 1982 and 1990, and they did not detect a significant change of this content in time.

For the deeper part of the lake, the results calculated with equation 4 are in agreement with observations (Figure 5). The model calculates a relatively small increase in sediment P during the first decade after creation of Lake Veluwe. From 1965 until 1978 the P pool increased rapidly and after the measures in 1979 it gradually decreased again. The calculated P content in 1998 is about half the maximum that was reached in 1978. According to the model, the P content will be below the initial content in 2006 and there will almost be an equilibrium between the supply to and the removal of P from the sediment top layer. The time period over which the sediment content

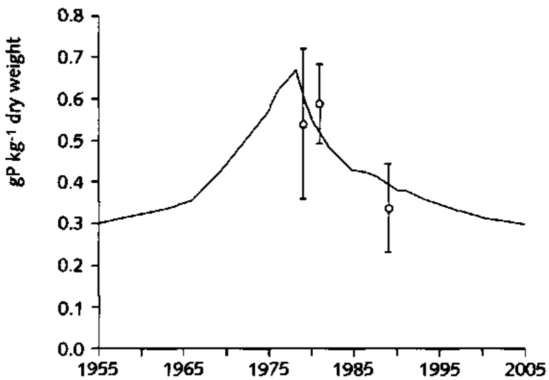


Figure 5 *Calculated P content of the sediment in the deeper part of Lake Veluwe and measurements (plus and minus standard deviation) on several locations in that area (1980 n=21, 1982 n=12, 1990 n=29).*

increased is comparable to the time period necessary to decrease the P content to its initial value. Also, the switch from clear to turbid water took place at a comparable level of the P content in the top layer of the sediments (1970, 0.44 gP kg^{-1} dry weight) as the switch from turbid to partly clear water (1990, 0.38 gP kg^{-1} dry weight).

The sensitivity of the calculated sediment P content in the deeper part of the lake in 1978 (maximum value) and 1998, and the year in which the initial content will be reached, was analysed for individual boundary conditions and model parameters (Table 2). No detailed information was available for the ranges of boundary conditions and model parameters. Therefore, the sensitivity was tested for a 10% increase of the nominal value; a decrease showed similar sensitivities. The sediment content in 1978 was mostly affected by the infiltration flux, the sediment density, the mixing depth and the surface area of the deep part of the lake. An increase in these parameters result in a decrease in the sediment P content in 1978. The calculated values for 1998 are insensitive for the sediment density, because an increase of the sediment density reduces both the effect of loading, seepage and infiltration per surface area. The sensitivities for the infiltration flux, the mixing depth and the area of the deep part are higher for 1998 than for 1978 and the burial rate becomes also more important. Since 1978, the addition of P to the sediment top layer has been lowered, which is reflected by the increased sensitivity for the burial rate and the infiltration flux.

The highest sensitivity of sediment P is found for the infiltration flux and the burial rate. The infiltration flux is the product of a flow rate, estimated with a model, and a concentration that is related to sediment P by a Langmuir equation and three parameters. The sensitivity of the sediment P pool for these parameters is comparable or even higher compared to the sensitivity for the total flux. The uncertainties in the values used for the infiltration flow rate, the parameters that relate the P concentration in the infiltrating water to sediment P, and the burial rate, are relatively high. Therefore, the effect of combined variations in the infiltration flux and the burial rate was analysed by assuming normal probability distributions for the infiltration flow (average value for the deeper part of the lake $90 \cdot 10^6$ m^3 y^{-1}, standard deviation estimated at $20 \cdot 10^6$ m^3 y^{-1}) and the burial rate

Table 2 *Changes in calculated sediment P content in 1978 and 1998, and the year in which the initial sediment P* *content of 1956 is reached again for the deeper part of Lake Veluwe, when the model parameters and input variables are increased individually with 10%.*

	% in 1978	% in 1998	Year
Initial sediment P composition (gP kg^{-1} dry weight)	0.1	0.0	
Mixing depth (m)	-1.9	2.3	2008
Burial rate (m y^{-1})	-1.0	-2.3	2003
Sediment density (kg dry weight m^{-3})	-2.9	-0.0	2005
Surface area deep part (m^2)	-1.9	2.3	2008
P erosion from shallow part (gP m^{-2} y^{-1})	0.2	0.6	2007
P infiltration flux (gP m^{-2} y^{-1})	-2.8	-5.1	2001

(average value for the deeper part of the lake $2.6 \cdot 10^{-3}$ m y^{-1}, standard deviation estimated at $1.0 \cdot 10^{-3}$ m y^{-1}). The resulting distribution of the sediment P content in 1998, based on 1000 model runs, has a median of 0.33 gP kg^{-1} dry weight and the 80% central confidence interval ranges from 0.27 to 0.40. Combinations of a low burial rate and a low infiltration flow resulted in a P content of 0.40 - 0.50 gP kg^{-1} dry weight. On the other hand, high values resulted in a P content less than 0.25 gP kg^{-1} dry weight. The sediment P content in 1998 was less than the initial P content in 30% of the model runs.

Another assumption that may affect the model results is the percentage retention of the external loading before 1976. A 10% increase of this value resulted in only 2.2% and 0.2% higher values for the P content in 1978 and 1998 respectively. The effect of an increased retention is partly balanced due to increased losses by infiltration.

A longer hydraulic residence time will result in a higher percentage retention. Additional calculations showed that an increase in the P retention from 46% to 70% from 1998 onwards, result in an elevated P pool in the sediment (Figure 6). Further, the effect of the recovery of macrophytes was simulated. The recovery of macrophytes in the shallow part of the lake has a dramatic effect on erosion and sedimentation in Lake Veluwe (Van den Berg *et al.*, 1998). The redistribution of P from the shallow part of the lake to the deeper part and consequently the burial in the deeper part of the lake, will be reduced. Figure 6 shows the effect of the development of macrophytes, assuming that from 1990 onwards 50% of the shallow area is covered with macrophytes, which then completely prevents erosion.

Discussion

The P concentration in the water is coupled with sediment P by internal P loading. In shallow lakes, this internal loading even frequently determines the concentration in the water during the growing season more than the external loading (Van der Molen & Boers, 1994). Moreover, the comparatively slow response to boundary conditions makes the P content of the sediment top layer a suitable variable to study long-term behaviour of shallow lakes. In terms of Figure 1, the wide range between line A and C would become narrower if the x-axis would be expressed in sediment P.

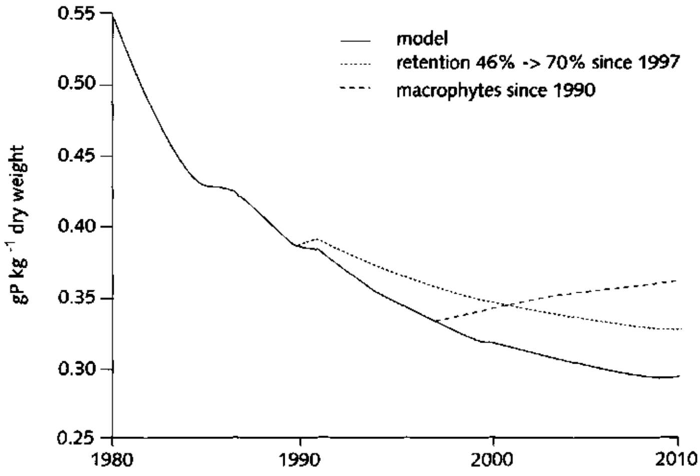


Figure 6 *The effect of the percentage P retention and the effect of macrophytes on the sediment P content of Lake Veluwe.*

A simple mass balance model for sediment P has been extended and applied to Lake Veluwe. All parameters and boundary conditions of the model are based on side-specific observations. However, several assumptions had to be made. Some observations are based on one or a few locations and had to be extrapolated to the whole lake. Further, in the first two decades since the creation of the lake the number of observations was limited. The sensitivity of the model results for the parameters and the boundary conditions has been tested. The sensitivity analysis illustrated that small variations in parameters and boundary conditions, compared to their confidence intervals, resulted in significant variations in the model output. The sensitivity expressed as percentage of the variation ranges from 0 to 29%, which is considered to be acceptable with respect to the intended use of the model. Only variation of the infiltration flux resulted in a larger variation of the sediment P content in 1998 (51%). Combined variation with an estimated, realistic standard deviation of the two most sensitive factors (infiltration and burial) resulted in a realistic and an acceptable interval of the model response.

The model describes the most important fluxes determining the sediment total P content. Therefore, the model is expected to be applicable at least to lakes with similar characteristics. For Lake Veluwe, the model results point out that the P content of the sediment in the deeper part of the lake is determined by (a) the retention of P, (b) infiltration of dissolved P to deeper sediment layers and (c) burial of the sediment (Figure 7).

(a) Both variation of the flushing rate and changes in the trophic state may affect the percentage retention of P. A longer residence time, as a consequence of reduced flushing, result in a higher age retention, but also in a lowered external loading. Hence, variation of the flushing of the lake will hardly affect the absolute level of P retention and changes in sediment P will be less than illustrated in Figure 6. P retention may increase with decreasing productivity, due to increasing benthic activity and enhanced adsorption of phosphate as a consequence of improved aerobic conditions in the sediments. Again, lower productivity caused by a decreased external loading will have little effect on the absolute level of P retention.

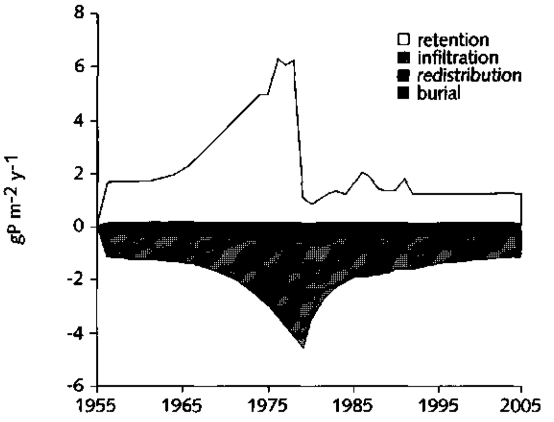


Figure 7 *P fluxes in and out the top layer of the sediments in the deeper part of Lake Veluwe.*

(b) The typical hydrology of the lake, with high infiltration towards the adjacent polder, is responsible for the relatively fast recovery of Lake Veluwe from the high external P loading in the past. The coupling of the P concentration in the infiltration water to sediment P results in a relatively slow build up of P in the sediments during periods with a high external loading.
(c) An important aspect of the application of the model to Lake Veluwe is the redistribution of solids from the shallow part to the deeper part of the lake. Burial with eroded material from the shallow part of the lake dominates burial by external suspended matter and in-lake formation of solids. The development of macrophytes in the shallow part of the lake affect this erosion and sedimentation pattern. The assumption that 50% coverage with macrophytes completely prevents erosion may results in an overestimation of the effect of macrophytes. The coverage is almost 50% only in the growing season. However, the P content of the sediment in the deeper part of the lake will increase due to a lower dilution with sandy material. This may be counteracted by a reduced sedimentation of P, as part of the retention will now take place in the shallow areas. The formation of an organic layer between the macrophytes has already been observed. The effect of the macrophytes may be that the sediment P pool increases mainly in the shallow part of the lake. However, with constant external conditions the lake averaged P content of the sediment will decrease, so it is expected that the ecological recovery of Lake Veluwe will proceed in the coming years.

Conclusions

– A mass balance model of total P over the top layer of the sediment is extended and applied to analyse and predict the P status of Lake Veluwe. The model results are sensitive to variations in all parameters and boundary conditions, so there is no redundancy in the model structure. According to the model, the P content of the sediment in the shallow part of the lake is constant, which is in agreement with measurements. The P content of the deeper part of the lake is described satisfactory by the model.

– The time scale of the decrease of sediment P to its initial level explains the 'hysteresis effect' that is frequently observed in shallow lakes that have been exposed to nutrient enrichment and amelioration. As the available P in the lake water is partly determined by the P pool in the top layer of the sediment, switches from clear water to turbid water or vice versa, will be retarded as compared to the variation in the external P loading. High infiltration rates may explain the relative fast response of Lake Veluwe compared to other lakes.

– For long-term predictions on shallow lake trophic conditions, information on changes of the P pool in the sediments is a prerequisite. Based on model results, a further recovery of Lake Veluwe may be expected under current boundary conditions. Changed conditions due to variations in lake flushing or the recovery of macrophytes will not have a negative effect on lake recovery.

References

Benndorf, J., 1987. Food web manipulation without nutrient control: a useful strategy in lake restoration? *Schweizerische Zeitschrift für Hydrologie* 49(2): 237-248.

Berger, C. & A. Bij de Vaate, 1983. Limnological studies on the eutrophication of Lake Wolderwijd, a shallow hypertrophic *Oscillatoria*-dominated lake in the Netherlands. *Schweizerische Zeitschrift für Hydrologie* 45: 458-479.

Berner, R.A., 1974. Kinetic models for the early diagenesis of nitrogen, sulfur, phosphorus and silicon in anoxic marine sediments. In: E.D. Goldberg (ed.), The Sea: Marine Chemistry, Vol. 5. John Wiley & Sons, New York, pp. 427-450.

Boers, P.C.M., J. Van der Does, M.P. Quaak & J.C. Van der Vlugt, 1994. Phosphorus fixation with iron(III)chloride: A new method to combat internal phosphorus loading in shallow lakes? *Archiv für Hydrobiologie* 129: 339-351.

Boström, B., G. Persson & B. Broberg, 1988. Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiologia* 170, 133-155.

Brinkman, A.G. & W. Van Raaphorst, 1986. De fosfaathuishouding in het Veluwemeer. Proefschrift Technische Hogeschool Twente, The Netherlands (in Dutch with an English summary).

Cooke, G.D., E.B. Welch, S.A. Peterson & P.R. Newroth, 1993. Restoration and Management of Lakes and Reservoirs. Boca Raton, Lewis Publishers.

Danen-Louwerse, H.J., L. Lijklema & M. Coenraats, 1993. Iron content of sediment and phosphate adsorption properties. *Hydrobiologia* 253: 311-317.

Danen-Louwerse, H.J., L. Lijklema & M. Coenraats, 1995. Coprecipitation of phosphate with calcium carbonate in Lake Veluwe. *Water Research* 29: 1781-1785.

Danen-Louwerse, H.J., R. Portielje & L. Lijklema, 1996. Phosphorus fixation in lake sediments. The case of Lake Veluwe. Agricultural University Wageningen, Department of Water Quality Management and Aquatic Ecology, report M274.

De Lange, W.J., 1996. Groundwater modeling of large domains with analytical elements. Thesis, University of Delft, The Netherlands.

Hosper, S.H., 1986. Restoration of Lake Veluwe, The Netherlands, by reduction of phosphorus loading and flushing. *Water Science and Technology* 17, 757-768.

Jagtman, E., D.T. Van der Molen & S. Vermij, 1992. The influence of flushing on nutrient dynamics, composition and densities of algae and transparency in Veluwemeer, The Netherlands. *Hydrobiologia* 233, 187-196.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer & K. Olrik, 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic temperate lakes 2: threshold levels, long-term stability and conclusions. *Hydrobiologia* 200/201, 219-227.

Jeppesen, E., P. Kristensen, J.P. Jensen, M. Søndergaard, E. Mortensen & T. Lauridsen, 1991. Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Memorie dell 'Istituto Italiano di Idrobiologie* 48: 127-148.

Jørgensen, S.E., L. Kamp-Nielsen & H.F. Mejer, 1982. Comparison of a simple and a complex sediment phosphorus model. *Ecological Modelling* 16: 99-124.

Kamp-Nielsen, L., 1983. Sediment-water exchange models. In: S.E. Jørgensen (ed.), Application of Ecological Modelling in Environmental Management, part A. Elsevier Scientific Publishing Company, Amsterdam, pp. 387-420.

Lijklema, L., 1982. A dynamic phosphate budget model for a eutrophic lake. *Hydrobiologia* 91: 227-233.

Meijer, M.-L., E.H. Van Nes, E.H.R.R. Lammens, R.D. Gulati, M.P. Grimm, J. Backx, P. Hollebeek, E.M. Blaauw & A.W. Breukelaar, 1994. The consequences of a drastic fish stock reduction in the large and shallow Lake Wolderwijd, The Netherlands. Can we understand what happened? *Hydrobiologia* 275/276: 31-42.

Noordhuis, R., 1997. Biologische monitoring zoete rijkswateren: watersysteemrapportage Randmeren. RIZA rapport 95.003, Lelystad (in Dutch with an English summary).

PER, 1986. Bestrijding van de eutrofiëring van het Veluwemeer-Drontermeer. Projectgroep Eutrofiëringsonderzoek randmeren, Rijkswaterstaat, Lelystad (in Dutch).

Scheffer, M., 1989. Alternative stable states in eutrophic shallow freshwater systems: a minimal model. *Hydrobiological Bulletin* 23: 73-85.

Smits, J.G.C. & D.T. Van der Molen, 1993. Application of SWITCH, a model for sediment-water exchange of nutrients, to Lake Veluwe in The Netherlands. *Hydrobiologia* 253: 281-300.

Van Ballegooijen, L. & D.T. Van der Molen, 1994. Water- en nutriëntenbalansen voor de randmeren van de Roggebotsluis tot de Nijkerkersluis over de periode 1976 - 1992. RIZA werkdocument 94.028X, Lelystad (in Dutch).

Van den Berg, M.S., H. Coops, M.-L. Meijer, M. Scheffer & J. Simons, 1998. Clear water associated with a dense Chara vegetation in the shallow and turbid lake Veluwemeer. *Ecological Studies* 131: 339-352.

Van der Molen, D.T. & P.C.M. Boers., 1992. Fosfaatbalans over de bodem van het Veluwemeer. RIZA werkdocument 92.020X, Lelystad (in Dutch).

Van der Molen, D.T. & P.C.M. Boers, 1994. Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. *Hydrobiologia* 275/276: 379-389.

Summary

In this thesis the role of eutrophication models in water management is analysed. The thesis consists of an extended introduction followed by five Appendices with papers describing different mathematical models dealing with eutrophication in surface waters. At first systems analysis is described as a general framework for eutrophication modelling. Emphasis is put on the uncertainties inherent in predictions. Next, a set of criteria is derived from this. The criteria may be useful in evaluating previous work and in guiding new initiatives on eutrophication modelling. Accordingly, the criteria are applied to review studies described in three of the five papers and to evaluate a number of projects of different lake managers in The Netherlands.

Systems analysis, an approach to solve problems, is utilised as the framework to study problems related to eutrophication of surface waters. The following stages are distinguished in this approach: problem analysis and the formulation of objectives, the set up of a conceptual framework, analysis of available data, specification of the mathematical formulations, sensitivity analysis, parameter estimation, validation and finally application of the model. Application of this approach generally results in repeatedly going through several stages, so systems analysis is a cyclic rather than a one-way procedure. Several examples from the field of eutrophication illustrate the approach.

Contrary to common practice of systems analysis the proper technical development and/or application of the model, referred to as 'credibility', is explicitly distinguished in this thesis from the acceptance of the results, referred to as 'acceptability'. This is in agreement with actual practice in water management, where the modeller, the user of the model and the user of the results of the model often are two or even more different persons. Credibility and acceptability are thus specified in a set of criteria which may be useful for both modellers and users of the model (results). The criteria addressing credibility are: the objectives of the model and the ensuing choice of the state variables, the dimensions of the modelled system and aggregations in time and space, the utilisation of the available data, the appropriateness of the model structure, the determination of parameter values, the validation of the model and finally the assessment of the uncertainties in the structure, the parameters and the results of the model. Criteria addressing acceptability are: attention to the motivation to undertake a modelling project, specification of the constraints in time and money, explication of the arguments to accept or reject (parts of) the model (results) and consideration of the consequences of the use of the model (results).

Next, the criteria are used to review three studies described in the Appendices. A statistical model aiming at predicting the phosphorus concentration in lakes is described in the first study. All available information was used to determine the structure and parameters of the model, so validation on another data set could not be carried out. However, recent studies indicated that the general conclusions from the study were correct. On the other hand, accurate prediction of the phosphorus concentration appeared to be unattainable.

A conceptual model for long term prediction of the phosphorus content in the top layer of the sediment is described in the second study. The parameter values were derived from previous work. Consequently, the model could be validated on observations of the sediments of Lake Veluwe, The Netherlands. The results of the validation were sufficient to qualitatively interpret and compare model predictions for a number of management alternatives.

In the third study a relatively complex model is presented. The model was validated on observations of Lake Veluwe as well. In this thesis the assumption of complete mixing of the water was confirmed. Nevertheless, spatial homogeneity of the sediment was questioned. Furthermore, the previous qualitative validation was quantified. A deviation of 10% - 40% between model results and observations was obtained for the main model variables and this was acceptable taking into account the inherent errors of the observations. Neither the parameters nor the model structure could unambiguously be identified. Furthermore, the validation was biased in so far some parameters were adjusted. Therefore, the model was rightly used for hypothesising and for speculating and not for comparing detailed alternative courses of action.

The assessment of uncertainties related to the use of these models did not always receive sufficient attention in the three studies discussed above. This thesis partly redeems the shortcomings and this contributes to the credibility of the studies. Next, subjects related to acceptability were highlighted. For that purpose lake managers at the operational level of three districts in The Netherlands were interviewed. Acceptability of a model or of the results of a model appeared to be only partly related to the credibility of a modelling project. Informal and personal relationships and accidental factors contributed significantly to the managers' choice of the model(ler). The acceptation of the results of a modelling project was also affected by the motivation of a manager to undertake a project and by the constraints in time and money imposed by the manager. In their turn, managers were willing to adjust their objectives and to pay little attention to the reliability of the results.

The constraints in time and money imposed by managers were the main reason why insufficient attention was paid to uncertainties in the development and the application of models. The assessment of uncertainties is laborious and complex, while completeness and objectivity are probably not feasible for larger models. That is why it is so important that the modellers clarify what is addressed and what is omitted. This, and not a high reliability in itself, is the essence of credibility. The managers are then better equipped to make verifiable choices to accept or reject (part of) the model (results). Managers are not always aware of the possible advantages of addressing uncertainties. Doing this may guide further actions, save time and money in certain situations, enhance public support and contribute to the confidence in the authority of the manager. Analogously to modellers with respect to credibility, managers may be more explicit in their arguments for accepting or rejecting (part of) the results. This may alleviate the present shortcomings and may contribute to a more transparent policy making and, thereby, further improve the role of eutrophication models in water management.

Summaries of the papers can be found in the Appendices.

Samenvatting

In dit proefschrift is de rol van eutrofiëringsmodellen in het waterbeheer geanalyseerd. Het proefschrift bestaat uit een uitgebreide inleiding gevolgd door een vijftal bijlagen met artikelen, waarin verschillende kwantitatieve wiskundige modellen voor de bestudering van de eutrofiëring van het oppervlaktewater centraal staan. Eerst is een algemeen kader gegeven met betrekking tot de modellering van eutrofiëring. Hierbij is speciale aandacht geschonken aan de onzekerheden die onlosmakelijk verbonden zijn aan het doen van voorspellingen. Dit kader is nader geconcretiseerd in een lijst met aandachtspunten, die gehanteerd kunnen worden bij de evaluatie van bestaand werk en als leidraad bij nieuwe ontwikkelingen met betrekking tot eutrofiëringsmodellen. Deze aandachtspunten zijn vervolgens toegepast voor de evaluatie van studies beschreven in drie van de vijf artikelen en van een aantal projecten bij regionale waterbeheerders in Nederland.

De systeemanalyse, een algemene methodiek om problemen op te lossen, is gebruikt als kader voor de bestudering van vraagstukken omtrent de eutrofiëring van het oppervlaktewater. Hierbij zijn de volgende fasen onderscheiden: probleemanalyse en formulering van doelstellingen, het opzetten van een conceptueel raamwerk, analyse van de beschikbare gegevens, het vaststellen van de wiskundige vergelijkingen, gevoeligheidsanalyse, schatting van de waarden van parameters, model validatie en tenslotte toepassing van het model. In de praktijk zal de methode niet recht toe recht aan kunnen worden gevolgd, maar zal een meer cyclisch karakter krijgen doordat bepaalde fasen soms opnieuw moeten worden doorlopen. De methodiek wordt geïllustreerd aan de hand van diverse voorbeelden op het gebied van eutrofiëring.

Anders dan gebruikelijk bij systeemanalyse is in dit proefschrift de correcte technische ontwikkeling en/of toepassing van het model ('credibility') onderscheiden van de acceptatie van het resultaat ('acceptability'). Dit sluit goed aan bij de huidige praktijk van het waterbeheer waar modelleur, gebruiker van het model en gebruiker van het resultaat van het model, vaak twee of meer verschillende personen zijn. 'Credibility' en 'acceptability' zijn hanteerbaar gemaakt door middel van een lijst met aandachtspunten, die van nut kunnen zijn voor zowel modelleur als gebruiker. Met betrekking tot 'credibility' betreffen deze aandachtspunten achtereenvolgens de doelstellingen van het model en de daaraan gerelateerde keuze van de toestandsvariabelen, de dimensies van het gemodelleerde systeem en de aggregatie in ruimte en tijd, het gebruik van de beschikbare gegevens, de juistheid van de structuur van het model, de keuze van de parameter waarden, de validatie en tenslotte de onzekerheden in de structuur, de parameters en de resultaten van het model. Met betrekking tot 'acceptability' betreffen de aandachtspunten de motivatie om een project te starten, de randvoorwaarden in tijd en geld die daarbij worden gesteld, het expliciet maken van de argumentatie om resultaten wel of niet te accepteren en tenslotte de consequenties van het gebruik van de resultaten.

De aandachtspunten zijn gebruikt voor de evaluatie van drie studies die zijn beschreven in de bijlagen. Een statistisch model dat de fosfor concentratie in een meer voorspelt is beschreven in de eerste studie. De beschikbare gegevens zijn allemaal gebruikt om de structuur en parameters van het model te bepalen, zodat er geen onafhankelijke data waren voor een validatie. Niettemin is uit recente studies gebleken dat de algemene conclusies getrokken op basis van de analyse juist zijn, maar dat een voldoende nauwkeurige berekening van de fosfor concentratie niet goed mogelijk is.

In de tweede studie is een conceptueel model beschreven voor de lange termijn ontwikkeling

van het fosfor gehalte in de toplaag van de waterbodem. De parameter waarden zijn ontleend aan voorgaande studies. Het model is gevalideerd op basis van enkele metingen aan de waterbodem van het Veluwemeer. De resultaten hiervan waren voldoende om voorspellingen met betrekking tot de effecten van enkele beheersopties kwalitatief te interpreteren en onderling te vergelijken.

In de derde studie is een relatief complex model gepresenteerd. Ook dit model is gevalideerd op gegevens van het Veluwemeer. In dit proefschrift is de aanname dat de water fase volledig is gemengd onderbouwd. Aan de andere kant is gesteld dat het discutabel is om uit te gaan van een horizontaal homogeen sediment. Verder is in dit proefschrift de voorheen kwalitatieve validatie van het model gekwantificeerd. Model resultaten en meetwaarden wijken 10% - 40% af voor de belangrijkste variabelen. Dit wordt acceptabel geacht gezien de fouten die inherent zijn aan de metingen. Echter, er is niet sprake van een echte validatie, omdat enkele parameters zijn bijgesteld. Ook bleek het niet mogelijk om de structuur en de parameters van het model eenduidig vast te stellen. Het model is daarom terecht slechts gebruikt voor de formulering van hypothesen en speculaties en niet voor vergelijking van gedetailleerde alternatieven.

Het omgaan met onzekerheden gerelateerd aan het gebruik van modellen heeft in bovengenoemde studies niet altijd voldoende aandacht gekregen. Dit is in dit proefschrift deels goedgemaakt. Hiermee is bijgedragen aan de 'credibility' van de studies. Vervolgens zijn aspecten belicht die meer te maken hebben met 'acceptability'. Hiertoe zijn waterbeheerders op het operationele niveau van een drietal regio's geïnterviewd. Het lijkt dat de acceptatie van een model of de resultaten daarvan maar ten dele samenhangt met de 'credibility' van de studie. Informele en persoonlijke relaties en toevallige factoren bepaalden voor een belangrijk deel de keuze van een waterbeheerder voor een model of de modelleur. De acceptatie van het resultaat werd verder beïnvloed door de aanleiding voor het project en door de randvoorwaarden (tijd en geld) van het project. Anderzijds bleken beheerders bereid hun doelen aan te passen en besteedden ze weinig expliciete aandacht aan de betrouwbaarheid van de resultaten.

De randvoorwaarden in tijd en geld, die door de waterbeheerders werden opgelegd, bleken de belangrijkste reden waarom er relatief weinig aandacht was voor de onzekerheden die een rol spelen bij de ontwikkeling en toepassing van de modellen. Het aangeven van de betrouwbaarheid van de resultaten is ook een omvangrijke, complexe taak, waarbij volledigheid en objectiviteit zeker bij grotere modellen waarschijnlijk niet kan worden bereikt. Het is juist daarom belangrijk dat de modelleur goed omschrijft wat wel en niet is gedaan op dit gebied. Dit, en niet zozeer een hoge betrouwbaarheid op zich, is de essentie van 'credibility'. De beheerder kan vervolgens een beter verifieerbare keuze maken tot acceptatie of verwerping van (een deel van) de resultaten. Beheerders zijn zich niet altijd bewust van de voordelen die aandacht voor de onzekerheden in het model of in de model resultaten op kan leveren. Het kan bevorderen dat meer gericht vervolg wordt gegeven aan het oplossen van het probleem, het kan in bepaalde situaties geld en tijd besparen, het kan het draagvlak vergroten en bijdragen aan de reputatie van de beheerder. Evenals voor de modelleur geldt ook voor de beheerder dat deze de argumenten voor het wel of niet accepteren van (een deel van) de resultaten zou moeten specificeren. Dit kan de tekortkomingen van de huidige praktijk beperken en bijdragen aan een meer doorzichtige besluitvorming en daarmee aan de verbetering van de rol van eutrofiëringsmodellen in het waterbeheer.

In bijlage 1, het artikel 'Environmental model calibration under different specifications: application to the model SED', worden diverse aspecten beschreven die een rol spelen bij de optimalisatie van (de parameters van) een model. Vooraf dient te worden nagedacht over wat men met het model wil bereiken: bijvoorbeeld een gemiddelde concentratie beschrijven, norm-overschrijdingen schatten of slechts een bepaald deel van de simulatie periode zo goed mogelijk beschrijven. Een unieke, beste set parameters bij een gegeven model structuur bestaat daarom niet. De consequenties van verschillende keuzen met betrekking tot het doel van het model worden geïllustreerd.

In bijlage 2, het artikel 'Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading', wordt met behulp van een statistisch model de fosfor concentratie in een meer voorspeld op basis van de gemeten externe- en interne fosfor belasting en de hydraulische verblijftijd. Het blijkt dat de externe fosfor belasting overwegend de fosfor concentratie in het meer bepaalt wanneer er geen maatregelen zijn genomen, maar dat de interne fosfor belasting een grotere invloed heeft op deze concentratie wanneer de externe belasting door maatregelen is verlaagd. Van bepaalde chemische eigenschappen van de waterbodem wordt verwacht dat ze beter te meten zijn en dat ze in de eerste jaren na een ingreep minder zullen veranderen in vergelijking met de interne fosfor belasting. Om te kunnen voorspellen wat de concentratie in het meer zal worden, na maatregelen gericht op verandering van de externe fosfor belasting en/of de hydraulische verblijftijd, is getracht de interne fosfor belasting als onafhankelijke variabele in het model te vervangen door deze eigenschappen. De correlatie tussen de berekende en gemeten fosfor concentratie bleef hoog, maar de voorspellende waarde werd beperkt door een tekort aan gegevens.

In bijlage 3, het artikel 'Mathematical modelling as a tool for management in eutrophication control of shallow lakes' wordt een relatief complex model gepresenteerd en toegepast op het Veluwemeer. Het model beschrijft niet alleen fosfor, maar ook bijvoorbeeld stikstof, silicium, chloride, algen (-soorten) en doorzicht. Hierdoor kunnen de vele metingen aan water systemen beter worden benut, maar neemt ook het aantal vergelijkingen en parameters sterk toe. De resultaten van het model komen in de meeste gevallen redelijk goed overeen met de metingen en op basis hiervan worden uitspraken gedaan over het functioneren van het systeem en over mogelijke effecten van bepaalde beheersmaatregelen.

In bijlage 4, het artikel 'Changes in phosphorus and nitrogen cycling following food web manipulations in a shallow Dutch lake' worden met behulp van een stationair model de fosfor- en stikstof hoeveelheden en stromen in het Wolderwijd beschreven voor twee verschillende jaren voor en vlak na een forse ingreep in het visbestand. Voor de ingreep zijn de meeste nutriënten gebonden in fytoplankton en vis. Na de ingreep is dat in respectievelijk de bentische algen, vis, zoobenthos en hogere waterplanten. Met de verwijdering van vis zijn ook veel nutriënten aan het systeem onttrokken, maar veranderingen in het systeem worden toegeschreven aan de geleidelijke verlaging van de externe fosfor belasting.

In bijlage 5, het artikel 'Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, The Netherlands', wordt een conceptueel model gepresenteerd waarmee het fosfor gehalte van de toplaag van de waterbodem wordt gesimuleerd. Aangenomen is dat dit gehalte een goede maat is voor de trofische toestand van het systeem zodat voorspellingen kunnen worden gedaan met betrekking tot de ontwikkeling van het watersysteem op langere termijn. Het model is toegepast op het Veluwemeer, dat aanvankelijk helder en plantenrijk was, later troebel en gedomineerd door algen en nu weer grotendeels helder en plantenrijk. Volgens de uitkomsten van het model wordt de heldere toestand van het systeem verder versterkt bij de huidige externe fosfor belasting en beheersstrategie.

Met dank aan

Het idee van een proefschrift en bijbehorende promotie is enkele jaren terug ontstaan. Eerder al hebben vele personen een basis gelegd waardoor het zover kon komen. Ten eerste wil ik mijn ouders danken voor de mogelijkheden die ze mij hebben geboden en de vrijheid die ze mij hebben gegeven bij de keuzes die ik heb gemaakt. Jammer dat papa er niet meer bij kan zijn.

Ik ben Harry van Huet en Tom Higgins erkentelijk voor hun opbouwende kritiek tijdens mijn stages. Jan Uunk ben ik zeer dankbaar voor het vertrouwen dat hij mij heeft gegeven tijdens het hoofdvak van mijn studie, bij de begeleiding van mijn vervangende dienstplicht en door mij te stimuleren om de functie bij DBW/RIZA te aanvaarden. Dankbaar ben ik ook de vele collega's, in tijdelijke en vaste dienst, binnen en buiten het RIZA, voor de prettige en stimulerende samenwerking. Jullie maken dat ik zo veel plezier heb in mijn werk! Van hen die later niet aan bod komen wil ik Leo van Ballegooijen noemen, omdat hij naast het werk ook altijd garant stond en staat voor (sportieve) afleiding, en Eric Marteijn en later Harry Hosper, omdat zij als bazen mij de mogelijkheid boden en stimuleerden dit proefschrift tot stand te brengen.

Met veel plezier denk ik terug aan de vele uitstapjes met Harry van Oosterhout. Heerlijk bijkomen door in weer en wind naar vogeltjes te kijken! Kees van der Guchte, Henk Wolters, Carolien van Houwen en Ruurd Noordhuis ben ik dankbaar voor vele avonden gastvrijheid en gezelligheid. Ik heb er van genoten en dat verzachte het gemis van mijn thuis in Wageningen en later Maastricht. Bij gezelligheid denk ik ook meteen aan vele andere vrienden en vriendinnen. Ik hoop nog veel met jullie te kunnen genieten!

Dan zijn er ook nog velen die hebben bijgedragen aan het proefschrift. Theo Claessen, Roelof Veeningen, Lucas-Jan Hooijkaas, Thea Helmerhorst, Liz van Duin, Alma van Oest, Jan Schoot Uiterkamp, Els Knaapen, Hans Los, Rob Portielje en afdeling Presentatie-RIZA, heel veel dank voor jullie advies, opmerkingen, discussie en andere inbreng. En dan zijn er natuurlijk de (co-)promotoren Jan Leentvaar, Paul Boers en Bert Lijklema! Jan, jouw instelling van 'als de regels ruimte laten doen we 't zoals we dat zelf willen' heeft mij altijd aangesproken. Zo kon ik gaan telewerken en zijn we als afdeling Algemeen Onderzoek naar Parijs afgereisd. Je hebt veel bijgedragen aan het geloof in eigen kunnen. Paul, het is met jou fantastisch samenwerken, zowel binnen het werk als daarbuiten. Kritisch, vol humor en niet geremd door 'hoe het hoort', ik hoop er nog veel van mee te maken. En Bert, dankzij jou heb ik me veel werk op de hals gehaald, maar dankzij jou heb ik er nu ook zo'n voldaan gevoel bij! Ik ben blij dat ik je mocht meemaken bij mijn studie, bij mijn eerste en tweede promotie onderzoek en bij het werk. Bedankt voor al je ideeën, suggesties en je geduld bij het corrigeren van al mijn onzorgvuldigheden!

Jeanne, bedankt voor de steun die je me gaf. Je hield me scherp en gaf me afleiding. Jeanne, Tjitse en Yke, jullie geven mij onbeschrijfelijk veel geluk en blijdschap.

Over de auteur

Het 'snoeken' in poldersloten op de vrije woensdagmiddag en later het vissen op snoekbaars in de Friese meren en kanalen legden de basis voor de interesse in water van Diederik Theo van der Molen (geboren in Singapore, 13 november 1961). Na het behalen van het VWO diploma aan het Nassau College te Heerenveen in 1980, begon hij aan de studie Milieuhygiëne aan de Landbouwuniversiteit te Wageningen. Zijn specialisatie lag bij de ecologische, fysisch-chemische en modelmatige aspecten van het waterkwaliteitsbeheer. Voor zijn studie had Diederik al op vrijwillige basis meegewerkt aan het water-onderzoek bij het Limnologisch Instituut dat toen nog deels was gevestigd in Oosterzee. Tijdens zijn studie heeft hij daar eveneens een stage uitgevoerd, waarbij hij als een van de eersten de gevolgen van de landbouwpraktijk voor de eutrofiëring van het oppervlaktewater in beeld bracht. Zijn tweede stage bestond uit het doen van onderzoek en geven van onderwijs in Ierland (Regional Technical College, Sligo). In 1987 behaalde hij het ingenieursdiploma, met als hoofdvak Waterzuivering en als bijvakken Natuurbehoud en Natuurbeheer en Informatica.

Daarna heeft hij de kans benut om nog meer van de wereld te zien. In zuidoost Azië heeft hij geleerd dat schoon water niet onbeperkt voor handen is en dat bewust omgaan met de voorraden een deugd is. Bij terugkomst kon Diederik direct beginnen met het vervullen van vervangende dienstplicht in DBW/RIZA te Lelystad. Hij werkte aan eutrofiëringsmodellen en dat bleek niet bepaald carrière onderbrekend. In 1989 kon hij beginnen als Onderzoeker In Opleiding aan de Landbouwuniversiteit te Wageningen. Het onderzoek richtte zich op de vastlegging van fosfaat in waterbodems. Vermeldenswaardig is verder dat hij dit in deeltijd deed, iets wat hij ook daarna is blijven doen. Dit promotie onderzoek heeft hij niet afgerond. In 1990 was de functie die hij als vervangende dienstplicht had vervuld in een vaste baan omgezet en uiteindelijk is hij overgestapt. Hiervan heeft hij nooit spijt gehad. De functie bestond uit werken aan waterkwaliteitsmodellen en meer in het algemeen aan de advisering van waterbeheerders en beleidsmakers. Het betrof zowel veld- en laboratoriumexperimenten als meer theoretisch werk. In de loop van de tijd ging meer en meer aandacht uit naar de bijdrage van de landbouwsector aan de nutriënten belasting van het oppervlaktewater. Diederik sprak regelmatig op congressen in binnen- en buitenland en er kwam een uitgebreide lijst met publicaties tot stand.

In de tussentijd zijn de namen van het instituut en de afdeling meerdere malen aangepast. Momenteel werkt hij bij de afdeling Water Systemen - Ecologie van het Rijksinstituut voor Integraal Zoetwatebeheer en Afvalwaterbehandeling (RIZA). Eind 1995 ontstond het idee voor dit proefschrift en het werk vond sindsdien deels plaats in tijd van Rijkswaterstaat. Per 1 september 1998 is hij programmaleider Ecologische Modellen en verantwoordelijk voor op peil houden van het instrumentarium waarmee de natte natuur kan worden beschreven. Diederik ziet dit proefschrift dan ook als een afronding van een voorgaande periode en tevens als een mooie start voor de komende tijd.

Lijst met publicaties

Van der Molen, D.T. & R. Portielje, 1999. Multi-lake studies in The Netherlands: trends in eutrophication. *Hydrobiologia*, in press.

Portielje, R. & D.T. Van der Molen, 1999. Relationships between eutrophication variables: from nutrient loading to transparency. *Hydrobiologia*, in press.

Van der Molen, D.T. & P.C.M. Boers, 1999. Eutrophication control in The Netherlands. *Hydrobiologia*, in press.

Van der Molen, D.T., R. Portielje, P.M. De Nobel & P.C.M. Boers, 1998. Nitrogen in Dutch freshwater lakes: trends and targets. *Environmental Pollution* 102 (S1): 553-558.

Van der Molen, D.T., R. Portielje, P.C.M. Boers & L. Lijklema, 1998. Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, The Netherlands. *Water Research* 32, 3281-3288.

Van der Molen, D.T., R. Portielje & S.P. Klapwijk, 1998. Vierde Eutrofiëringsenquête Meren en Plassen. *H_2O* 31, 11: 16-22.

Van der Molen, D.T., 1998. Bodemalgen in de randmeren; voorkomen en effecten op de nutriëntenhuishouding. *Diatomededelingen* 22: 10-15.

Van der Molen, D.T., R. Kruijne & E.J.B. Uunk, 1998. Verwijdering van stikstof en fosfor door bufferstroken langs de Mosbeek. *H_2O* 31, 3: 26-28.

Van der Molen, D.T., R. Kruijne & E.J.B. Uunk, 1998. Verwijdering van stikstof en fosfor door bufferstroken langs de Mosbeek. *Stromingen* 4: 27-40.

Van der Molen, D.T., A. Breeuwsma & P.C.M. Boers, 1998. Agricultural nutrient losses to surface water in The Netherlands. *Journal of Environmental Quality* 27: 4-11.

Portielje, R. & D.T. Van der Molen, 1998. Trend-analysis of eutrophication variables in lakes in The Netherlands. *Water Science and Technology* 37: 235-240.

Reeders, H.H., P.C.M. Boers, D.T. Van der Molen & T.H. Helmerhorst, 1998. Cyanobacterial dominance in the lakes Veluwemeer and Wolderwijd, The Netherlands. *Water Science and Technology* 37: 85-92.

Boers, P.C.M., W. Van Raaphorst & D.T. Van der Molen, 1998. Phosphorus retention in sediments. *Water Science and Technology* 37: 31-39.

Van der Molen, D.T. & P.C.M. Boers, 1997. Eutrofiëring en oligotrofiëring van het Veluwemeer. *Levende Natuur* 98, 1: 2-7.

Van der Molen, D.T., J.H. Janse & I. De Vries, 1996. Eutrofiëringsmodellen: overmaat schaadt? *H_2O* 29, 2: 34-37.

Michielsen, B.F., F.J. Los & D.T. Van der Molen, 1996. Modellering van eutrofiëring: toepassing op het beheer van het Veluwemeer van DELWAQ-BLOOM-SWITCH. *H_2O* 29, 12: 361-364.

Van der Molen, D.T. & P.C.M. Boers, 1996. Changes in phosphorus and nitrogen cycling following food web manipulations in a shallow Dutch lake. *Freshwater Biology* 35: 189-202.

Van Luijn, F., D.T. Van der Molen, P.C.M. Boers & W. Luttmer, 1995. The role of benthic algae in nutrient dynamics of shallow lakes. *Water Science and Technology* 32: 89-97.

Van der Molen, D.T. & A. Breeuwsma, 1995. Nutriëntenverliezen in de landbouw. Kloof tussen onvermijdbare en acceptabele nutriëntenverliezen. *Landschap* 6: 7-14.

Schoumans, O.F., R. Kruijne & D.T. Van der Molen, 1995. Vermindering fosfaatuitspoeling.

Mogelijkheden bij fosfaatverzadigde zandgronden. *Landschap* 6: 63-73.

Boers, P.C.M., I.H.S. Hoek, F. Klijn, M.A. Menke, A.F.M. Meuleman & D.T. Van der Molen (redactie), 1995. 'Vermesting' van oppervlaktewater. *Landschap* 6.

Van der Molen, D.T., F.J. Los, L. Van Ballegooijen & M.P. Van der Vat, 1994. Mathematical modelling as a tool for management in eutrophication control of shallow lakes. *Hydrobiologia* 275/276: 479-492.

Van der Molen, D.T. & P.C.M. Boers, 1994. Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. *Hydrobiologia* 275/276: 379-389.

Smits, J.G.C. & D.T. Van der Molen, 1993. Application of SWITCH, a model for sediment water exchange of nutrients, to Lake Veluwe in The Netherlands. *Hydrobiologia* 253: 281-300.

Van der Molen, D.T. & J. Pintér, 1993. Environmental model calibration under different specifications: application to the model SED. *Ecological Modelling* 68: 1-19.

Boers, P.C.M., J. Van der Does, D.T. Van der Molen & W.J. Rip, 1993. Eutrophication control in the Netherlands: approach, results, challanges for the future. In: G. Giussani & C. Callieri (eds.), Strategies for lake ecosystems beyond 2000, pp. 50-53.

Boers, P.C.M. & D.T. Van der Molen, 1993. Eutrophication control in Europe - the state of the art. *European Water Pollution Control* 3, 2: 19-25.

Boers, P.C.M. & D.T. Van der Molen, 1993. Wat kunnen we verwachten van een vermindering van de fosfaatbelasting op meren? H_2O 26, 26: 788-791.

Jagtman, E., D.T. Van der Molen & S. Vermij, 1992. The influence of flushing on nutrient dynamics, composition and densities of algae and transparency in Veluwemeer, The Netherlands. *Hydrobiologia* 233: 187-196.

Van der Molen, D.T., 1991. A simple, dynamical model for the simulation of the release of phosphorus from sediments in shallow, eutrophic systems. *Water Research* 25: 737-744.

Van der Molen, D.T. & T.H. Helmerhorst, 1991. Algen op de bodem van de randmeren. H_2O 24, 25: 719-724.

Van der Molen, D.T., P.C.M. Boers & L. Lijklema, 1989. Gegevens-analyse nutriënten en zwevend materiaal Veluwemeer. H_2O 22, 24: 760-762.