EUTROPHICATION RESEARCH

28-29 August 1997

WAGENINGEN, THE NETHERLANDS

Organised by:
- Department of Water Quality Management and Aquatic Ecology, Wageningen Agricultural University
- Netherlands research School for the socio-Economic and Natural Sciences of the Environment (SENSE)
- International association on Water Quality (IAWQ), Eutrophication specialist group
- Dutch Association for Water Management (NVA)
- Institute for Inland Water Management and Waste Water Treatment (RIZA)

Sponsored by:
- WAU, Wageningen Agricultural University
- RIZA, Institute for Inland Water Management and Waste Water Treatment
- WIMEK, Wageningen Institute for Environment and Climate Research
- SENSE, Netherlands research School for the socio-Economic and Natural Sciences of the Environment

preprints

edited by: R. Roijackers
R.H. Aalderink
G. Blom

Specialist Symposium
dedicated to
Lambertus Lijklema

EUTROPHICATION RESEARCH

STATE-OF-THE-ART

Inputs - Processes - Effects - Modelling - Management

28-29 August 1997
Wageningen, The Netehrlands

ORGANISING COMMITTEE

R.H. Aalderink[1]
Dr. P.C.M. Boers[2]
G. Blom[1]
Prof. Dr. L. Mur[3]
Dr. R.M.M. Roijackers[1]
P.T.J.C. van Rooy[4]

[1] Wageningen Agricultural University
[2] RIZA, Institute for Inland Water Management and Waste Water Treatment
[3] IAWQ, Eutrophication specialist group
[4] NVA, Dutch Association for Water Management

TABLE OF CONTENTS

TABLE OF CONTENTS (continued)

Management and restoration

Plenary lectures

Posters

Management and policies

Plenary lectures

Posters

TABLE OF CONTENTS (continued)

Effects

TABLE OF CONTENTS (continued)

INPUTS

THE CHALLENGE OF MANAGING WATER AND MATERIAL BALANCES IN RELATION TO EUTROPHICATION

P. Harremoës

Department of Environmental Science and Engineering
Technical University of Denmark, 2800 Lyngby, Denmark

ABSTRACT

Eutrophication is the result of overfertilisation of the aquatic environment. In other words, the detrimental effects derived from increased anthropogenic release of nutrients to the environment. By modern concepts of environmental abatement this is no longer only a question of identifying and purifying discharges. It is a question of keeping track of the system of nutrient mining, utilisation, recycle, transport, conversion, containment and release in the modern society - in effect, the total balance of nutrients in society. In Denmark, the point sources of nutrient release have been reduced by an order of magnitude. That has contributed to some improvements, but the diffuse sources have turned out to be more significant than originally expected. They are harder to reduce. That is why more integrated approaches have to be analysed. Four approaches will be analysed: the DPSIR-approach, the 5-options approach, the cause-effect relationship and the material balance analysis.

KEYWORDS

Eutrophication, nutrient balance, DPSIR, 5-options approach, wastewater treatment, urban runoff.

SETTING THE STAGE

Eutrophication is a good example of a problem, which requires a multi-disciplinary, multi-sectorial and multi-focal approach in order to reach abatement of the devastation created by the development of society during the last century. It started out with nuisance algae growth in lakes under heavy load from domestic waste-water. The problem was "solved" by adding a selective poison to the lake (Cubbersulphate) to remove the blue-green algae, 1930's. Then it was realised that the load of Phosphorus was the cause and the world became divided into religions with respect to measures by treating the wastewater or banning P in detergents during the 1950's and 1960's. Then it was realised that Nitrogen was limiting in estuarine and coastal waters and the world became divided into believers in P versus N as the limiting factor. In 1987 the Danish Government made a far-reaching decision: In view of the fact that the experts could not agree as to which nutrient is or could be made the limiting factor and in view of a political situation that called for action, it was decided to limit the pressure on the aquatic environment by reduction of the loads from Denmark of both P and N: P by 80% and N by 50%. This is an example of application of the precautionary principle: the uncertainty calls for more preventive measures than can be proven to be effective. The basic principle should

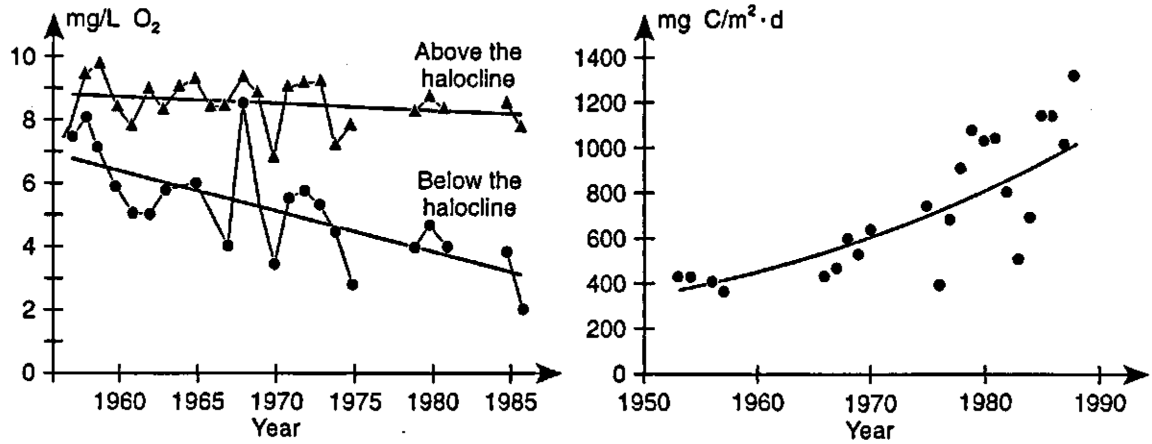


Figure 1: Development of oxygen concentration and C-production in the Danish marine waters over the last 30-40 years, from Hansen (1991) and Richardson and Ærtebjerg (1991)

be that the potential damage by more intensive measures than can be proven effective is less of a risk than the potential damage caused by doing too little, too late.

In Denmark, the reduction of the P-load has been successfully accomplished and the reduction of N from wastewater has been achieved; but the reduction from the diffuse sources has not been very successful. The debate as to whether P or N can most effectively be made the limiting factor will be with us for some time. The debate as to whether the precautionary principle is justly applied is still open, because the influence on something as fundamental as food production is at stake. All these issues call for the comprehensive analysis of the totality of the problem, accounting for all aspects of society. The approach is integrated environmental management.

THE DPSIR-ANALYSIS

Within the professional field of water pollution the rationale for abatement can be illustrated by the approach shown in figure 2. The rationale can be described as follows: Set the water ecosystem objectives by a political process based on identified beneficial water usage. Determine from this the water ecosystem criteria, which is a set of scientifically based relations between water quality and probability of certain effects. On this basis water ecosystem standards are set. The relationship between load and water quality is determined for the receiving water in question and from this the required load reduction can be calculated. It is analysed what remediating measures will achieve the required reduction of load and the cost of these measures is then weighed against the objectives with the political question: Does the objectives justify the costs. If not, a new set of objectives will have to be tried out and the right combination be found by iteration.

In the 1970's it was attempted to perform this approach on a theoretical basis: **The modelling approach**, i.e. on the basis of models for all elements in the system. Historically, it has to be acknowledged that the system did not work well in practise. The process was too slow to meet the political demand for action. The models were too complicated and inaccurate. The alternative has been **the iterative approach.** Some measures were decided, implemented, analysed and found inadequate for achieving the water ecosystem objectives and then after some years new measures were introduced to improve the situation. In this process both theoretical and purely empirical considerations have influenced the decision making. An example of this is the decision in Denmark to damand wastewater treatment from all houses not connected to a sewer system; essentially rural

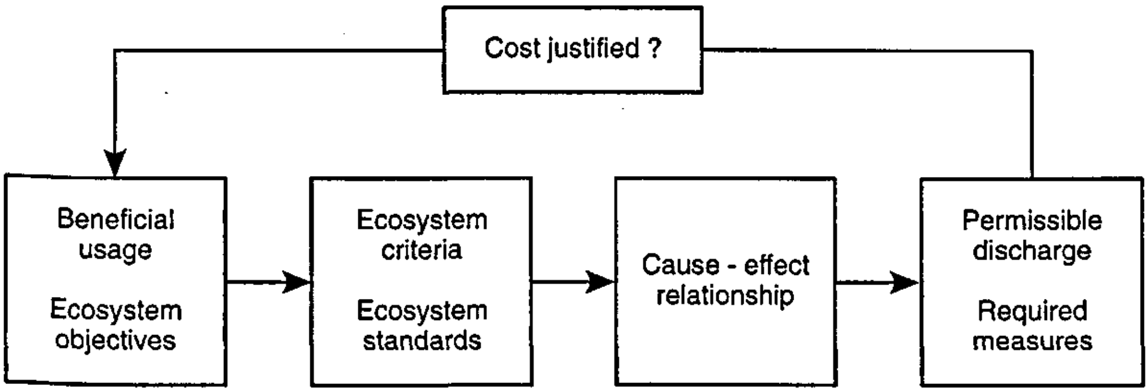


Figure 2: The traditional approach to abatement of water pollution, dating from the 1960'es

homes. This was based on the experience that up-stream rivers did not improve in quality on the basis of the general wastewater treatment scheme.

An approach to universal formulation of integrated environmental management of the so-called **DPSIR**-approach, accounting for the **D**riving forces in society, the **P**ressures on the environment caused by these driving forces, the **S**tate of the environment as a consequence of these pressures and the **I**mpacts imposed by that state and finally the **R**esponses of society with respect to measures with which to remedy the situation. The approach is illustrated in figure 2. The approach is derived from an air pollution context, but incorporates all aspects, including air, water and soil, and the basic principle is the same as the original approach from the water sector.

In relation to eutrophication the five categories can be identified as follows:

Driving forces:	The basic driving forces are realted to increase in population, increasing living standards based on abundant food production and the concentration of population in cities with the sewer system, traffic and energy production, contribution to the release of nutrients. Wat is the future of these needs?
Pressures:	Agriculture needs fertilizers, some of which peculates to the aquatic environment, manure from husbandry leads to NHx release to the air, urban development included sewerage resulting in discharge of nutrients, traffic releases Nox to the air.
State:	Overfertilisation leads to excessive algae blooms, high concentrations of algae, change of the aquatic ecosystem.
Impacts:	Deteriorating ecosystem to an undesirable state, poor fisheries, decreased amenities.
Responses:	Remediating measures to reduce the pressure on the eutrophied waters or accept the reduced state of the environment.

Figure 3 also illustrates the responses available for the society in curbing with a particular situation. This involves a number of terms from the political scene on environment:

Indicators:	Most environmental issues are very complicated. Eutrophication is no exception. In order to communicate with the public and the political system well identified simplifications are needed. These simple parameters are called indicators. They are symbolic of the pollution and hopefully well related to the real issues.

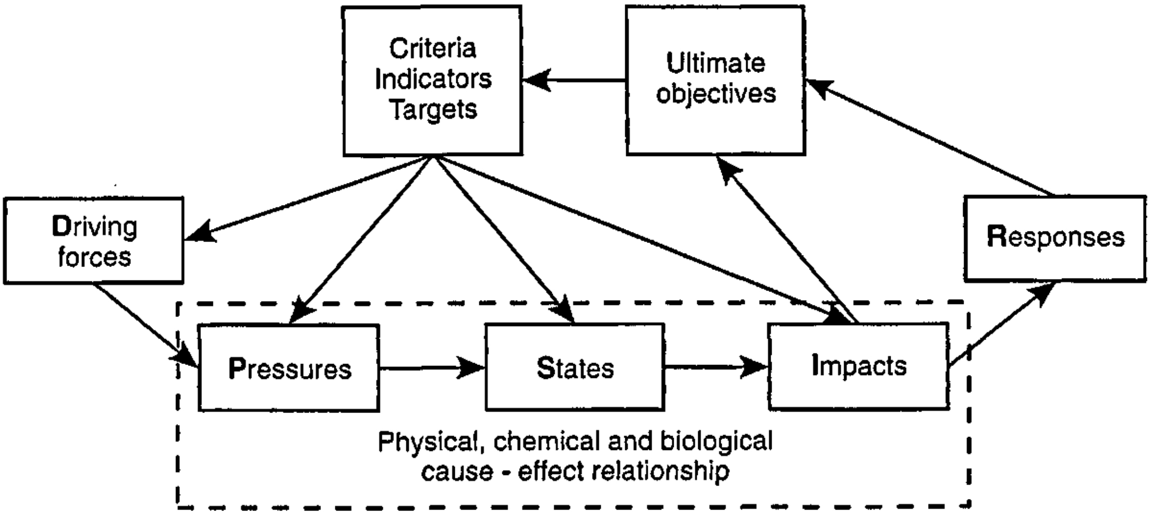


Figure 3: Illustration of the DPSIR-concept for management of environmental problems

Targets: By the same concept the political system has to set goals for environmental abatement in simple terms for everybody to understand. They are called targets and include, what an indicator should be in a few years time on the basis of the policies to be implemented.

Implementation: Express the tools chosen as the means by which to implement the policies for reaching a target.

As an example related to eutrophication the visibility in a lake (measured as secchi disk reading) may be chosen as indicator. One meter visibility may be the target and reduction of the P-load by wastewater treatment down to 0.1 mg/l in the effluent from all wastewater discharges in the catchment may be the measures by which to reach this before year 2000.

CAUSE-EFFECT RELATIONSHIPS.

Hidden within this concept are very complicated scientific relationships. The relationship between pressures, states and impacts involves extremely complicated physical, chemical and biological cause-effect relationships, which the environmental sciences have worked on for decades. It has to be acknowledged that there is a need for the simplifications in the form of indicators and targets as a means for communications with the public and the political system; but at the same time it has to be acknowledged that there is a danger in the distraction of the issues from the complicated reality. Failures due to misconceptions with respect to the cause-effect relationships can cause environmental as well as political disasters.

The approach to description of the cause-effect relationships has been deterministic mathematical models, developed over the last three decades. The basis for the development has been the concept that description of every physical, chemical and biological process in detail in an integrated mathematical framework will in summation describe the complicated nature of the cause-effect relationship. This is called the reductionist approach. Even in the description of rather simple physical systems the uncertainty and the sensitivity of these models is significant. In the description of complicated biological systems the reductionist approach has experienced failures. Eutrophication of lakes is an example. The level of eutrophication is not only determined "from below" by the load of nutrients on the system giving rise to high concentrations of algae, but it is also controlled "from above" by the function of the food web. There may be several states of equilibrium (or several attractors in a chaotic system) and the system can be trickered to shift from one state to another. That means that a high load of nutrients on a lake can lead to either a thick soup of algae or to a very high rate of fish production. How to master these changes can be a real challenge, even in simple systems like sewage ponds

It is frequently forgotten that if the deterministic models do not incorporate the most essential processes they may fail - irrespectively of the detail in the description of the other processes. Rough holistic models may be better than reductionistic models, in spite of the holistic models may suffer from many omissions of detail. The way forward is to make simple, but adequate models which incorporates the uncertainty in the mathematical framework (stochastic models, grey box models).

In this debate pro et con models it must not be forgotten that some form of understanding and description of the whole chain of the cause-effect relationships from pressure to impacts has to be the basis for proper decision making. The alternative is either political opportunism, environmental religion or guesswork.

It is a real challenge how to apply the precautionary principle in cases where the cause-effect relationships are either vaguely known or known but very uncertain. There is a whole discipline still to be developed.

THE 5-OPTIONS

Assuming that there is a unique relationship between the pressure in the form of a load on the environment and the impact there are only 5 options to consider with respect to abatement. These options can be listed as follows: Consider a chemical that is used in society: What options with respect to ultimate fate are available?

The 5-options concept is illustrated in figure 4.

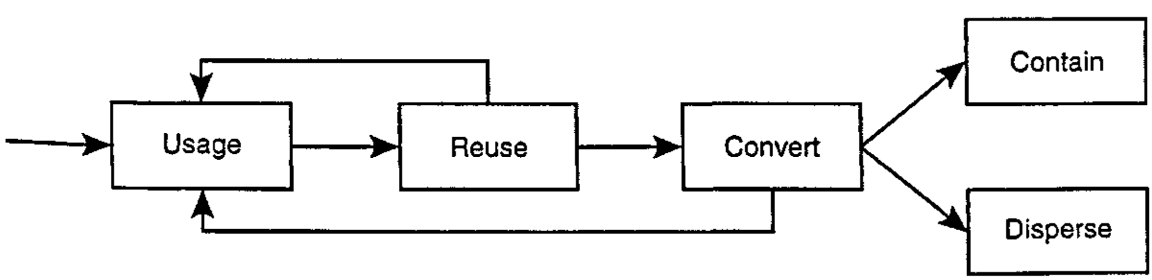


Figure 4: The simple concept of the 5 options available to abated management of pollution of the environment with chemicals. No other options are available and the choices have to be made

No use: We can stop using an unwanted substance, because the advantages of its use are overshadowed by the detrimental effects to the environment.

The historical example is the hard detergents in washing powder, which were banned in Germany in the early 1960'es. Many examples have been added since (DDT, PCB, etc.). The key is to find environmentally sound substitutes.

Reuse: We can decrease the amount reaching the environment by internally reusing the substances.

That is a very viable solution, but it has to be realized that there will still be a residue to be considered.

Convert: When a substance has been introduced it is important to control the route of that substance such that the transport can be identified and the flow treated. Treatment mostly means converting the substance from an obnoxious form to a form, acceptable for further transport by air or water or in solid form.

Conversion by incineration converts solid matter into an inert solid form and into a gaseous form. Wastewater treatment has the function to convert into a separable solid form or into a gaseous form. The essence in this context is that the solution is rarely a final one. It is just a conversion into something that can be transported acceptably on the next route.

Contain: One of the ultimate fates is to contain the residues and leave it there forever.

The deposit of radioactive waste in old salt mines is the best example. The material is deposited in the hope that it will never seep, creep, leach or migrate to the environment. The difficulty of this notion is illustrated by the landfill, which supposedly was aimed at as an ultimate solution. Leakage from landfills has been identified as an essential problem that calls for treatment. By modern concept the landfill is a treatment unit for many years until ultimately an inert residue has found its final resting place (until archaeologists find interest in the remains with the aim to learn about our behaviour).

Disperse: The only other ultimate fate is to disperse in the environment.

The gas from incineration has to be treated such that the residues in the flue gas are acceptable from an environmental point of view. The wastewater treatment shall create effluents with residues acceptable to the aquatic environment. The treatment of solid waste shall be such that the product can be either recircled or dispersed or contained.

There are no other options! (except shooting into space).

The system has to be looked at in total. Water can only be considered within the context of mass balances in societies. Flow of matter in society can no longer be looked at except in the total context of resources, society and environment. Cradle to grave, input-output and mass balance considerations have become indispensable tools. The instruments are available, though still under development and refinement.

In relation to eutrophication the ban of P in detergents is a good example of "no use". The alternative is to "convert" the P from soluble PO4 in the wastewater to insoluble P in sludge by chemical precipitation or by enhanced bio-P-uptake. It is an interesting fact developed over the last three decades that both approaches have been implemented; in spite of the fact that removal from the treatment plant could have achieved the same result without reformulation of detergents. P-free detergents have become a sales promotion.

No matter how well the water is treated there will always be a residue in the effluent to be "dispersed" in the environment. However, the concentration can be brought down to any low concentration. It is a question of cost versus benefit. There is nothing called Best Available Technology (BAT). There is always a better technology and it is available if the need arises. There will always be a waste product in the form of sludge. The solution to the nutrient discharge creates a new problem. The environmentally best solution is to "reuse" the P-content of the sludge as a fertiliser in agriculture; but this approach is marred with the risk of sludge contamination from other pollutants: metals and special refractory organiscs. The alternative is that the sludge is brought directly or via incineration to "containment" in landfills.

POLITICAL INSTRUMENTS

There are a number of political instruments available for implementation of policies:

Rules: Legislation, directives, codes, norms, etc. has been the traditional approach to regulation of pressures on water quality. The approaches vary:
- water ecosystem quality standards
- permissible effluent concentrations
- required treatment technology
- ban of specific chemicals

In Denmark, all these approaches are being applied in combination

Economy: Introduction of taxis, levies, etc. have been used in the past, such as the cost of discharging BOD from industry to the sewer system. However, introduction of economic incentives is getting increasing attention as an alternative to rules. It has greater flexibility, but it may also distort the rational basis for the approach to implementation.

Attitudes: The most effective (and the cheapest) approach to implementation is by change of attitude. It may involve the public and the professionals. The development over the last century of general hygiene is a successful example, performed every day by every person and in every kitchen without questioning the rational for the approach.

Rules can successfully be applied to point sources, in relation to eutrophication illustrated by the rules for wastewater treatment. However, rules are not suited for diffuse sources. Diffuse sources increase in relative importance and the other two alternatives will play an increasing role. Change of attitude is a slow process, frequently too slow for the political demand for action. However, in relation to regulation of agriculture it should be realised that there will be no successful solution without a basic change of attitude among farmers as to how to go about the daily operation of the farm. In that regard, the tough competition and the distortions created by the EU agricultural policies is detrimental to a change of attitude. Therefore, economic regulation will dominate the future regulation of pressures on the environment from agriculture.

Table 1: Contribution of Nitrogen to the Danish inland marine waters, in tons N per year in 1994

Types	Danish sources	Types	Other sources
Agriculture	87000	Air transport	29000
Treatment plants	9100		
Natural areas	3000	Net from the Baltic Sea	31000
Industry	1850	Net to the North Sea	-28000
Traffic	1500		
Non-sewered housing	1000	Germany	20000
Urban storm drainage	1000	Sweden	45000
Power plants	1000		
Fish ponds	825		
Total	**106275**	**Total**	**97000**

Sources: Miljø- og Energiministeriet, DMU (1996)

THE NUTRIENT CONTRIBUTIONS

Table 1 illustrates the contributions of nitrogen from Denmark and from other countries. The table illustrates the well known fact that the contribution of nitrogen from agriculture is far greater than the contribution from domestic wastewater. That fact has been well established in Denmark since the mid 1970's. In spite of that the Danish parliament decided to reduce the nitrogen load by 50% from all sources. It is still argued that this was an error of judgement. The argument against was based on the opinion that reduction in agriculture would be much cheaper than from wastewater. The argument in favour was based on the concept that it would be impossible to persuade agriculture to reduce their contribution unless the urban point sources were reduced equally. It has to be remembered that agriculture at the time argued that the domestic sources were the culprits and the P was the most likely limiting factor.

Under pressure in the political process agriculture argued that they had the tools by which to reduce their contribution. That happened by introduction of **Best Management Practise**. This involved green fields all year round, limitation of husbandry per hectare, limitations on application of manure and sludge; but it did not involve any limitation on the application of artificial fertilisers. The changes to agricultural practise have been quite significant, but the fact is that the measures have not lived up to the promisees made at the time. Ten years later, the mass reduction of nitrogen to the environment is no more than 20%.

It is interesting to analyse the interpretation of the word **"Best"** in the concepts "BAT" and "BMP". The political system has difficulty deciding anything but the general principle and rely on the professional system for advice. The professional system apply the word "Best" in order to advance their approach and interpretation of standard, because: Which politician does not want to do "the best". The fact is that it is not the best, because there is always something better. This is where the professionals come under pressure to make political interpretations and end up playing the role of political manipulators.

THE URBAN CONTRIBUTIONS

The dominant urban source of nutrients is domestic wastewater discharge, see table 1. It has been clearly demonstrated that the contribution from combined sewer overflows and separate system outlets during rain is a minor importance from a mass loading point of view, Harremoës (1989) and Harremoës and Sieker (1993).

Nitrogen

The decision of the Danish Parliament in 1987 to reduce all discharges by 50% was implemented by enforcing an effluent concentration less than 8 mg/L as a mean mass discharge for all treatment. The technical approach is nitrification-denitrification. In the period 1985-95 the treatment plant contribution was reduced by 68%, Christensen (1996). In 1998 the implementation scheme has been successfully achieved with the two big treatment plants of the City of Copenhagen as the last now in operation, Harremoës, Sinkjær and Hansen (1992).

The same development will take place in Europe on the basis of the EU wastewater directive.

From a technical point of view this development has been very satisfactory. Nitrification-denitrification has been the research and development issue at The Department of Environmental Science and Technology for more than 25 years, Harremoës, Bundgaard and Henze (1991) and Henze, Grady, Gujer, Marais and Matsuo (1987). In the beginning the claim was that nitrogen removal was not needed, not feasible and if feasible too expensive. When it was needed the technology was ready which illustrates the meaninglessness of the concept of "Best Available Technology". Nitrification-denitrification was in fact "available" since the mid 1970's.

Phosphorus

Removal of P is based on two approaches: chemical precipitation and enhanced bio-P-uptake. Chemical treatment has been available since the late 1960's, while bio-P-removal has been available but not sufficiently understood to guarantee performance until recently. The removal of P from detergents and the consequential reduction in the inlet concentration to the treatment plants and cost works in favour of bio-P-removal, but it is still not quite reliable and no treatment with bio-P-removal should be without chemical treatment facilities as a back-up.

The 1987 implementation of 80% removal was manifested as a required effluent concentration less than 1.5 mg/L. In the period 1985-95 a reduction by 79% was achieved, Christensen (1996), to be further reduced when the two be treatment plantsof the City of Copenhagen in in full operation by mid 1997. However, thisP-load will be decreased further, because recently in Denmark a new discharge tax on P has been implemented and the tax has been set such that it is economical to treat to concentrations in the order of 0.2-0.4 mg/L. That will favour chemical precipitation, but a combination of bio-P-removal and chemical precipitation is foreseen as the most likely scenario for the future.

CONCLUSIONS

The regulation of nutrient discharges to the aquatic environment can be achieved by settings rules for permissible discharges. The technology is available for the current formulations of effluent standards. If these standards turn out not to achieve the desired environmental standard the effluent standards can be strengthened by an iterative approach. The technology is available for improvement. However, the problems with eutrophication abatement will be associated with diffuse sources of nutrients. The control of nutrients from agriculture will require a much more integrated and comprehensive approach and an analysis of the options available for implementation. The approaches will include rules for agricultural behaviour, economic taxation and/or incentives and long range change of attitudes.

REFERENCES

Christensen, T. Miljøstyrelsen, personal communication (1996).

Hansen, I.S. (1991) Transport af vand og stof inden for og til Kattegat og Bælthavet - herunder ændringer siden 1950'erne. Konsessuskonference 1991, kap. 15

Miljø- og Energiministeriet, DMU 1996/9, Kvælstofbalance for havmiljøet.

Harremoës P. (1989). Overflow quantity, quality and receiving water impact. In: Urban discharges and receiving water quality impacts, Pergamon Press, Oxford, UK.

Harremoës P., Bundgaard, E. and Henze M. (1991) Developments in wastewater treatment for nutrient removal, Wat. Poll. Contr. 1, 19-23.

Harremoës P., Sinkjær O. and Hansen J.L. (1992) Evaluation of methods for nitrogen and phosphorus control in sewage effluents, J. IWEM, 6, 50-61.

Harremoës P. and Sieker F. (1993). Influence of stormwater storage tanks on pollutant discharge to receiving waters. 9th EWPCA-Symposium, Munich.

Henze M., Grady, Jr. C.P.L., Gujer W., Marais G.v.R. and Matsuo T. (1987) Activated sludge model no. 1, Tech. Rep. no. 1, IAWPRC, London

Richardson, K. and Ærtebjerg, G. (1991) Effekter af kvælstof og fosfor i Kattegat of Bælthavet. Konsessuskonference 1991, kap. 16

NITROGEN AND PHOSPHORUS LOSSES FROM AGRICULTURE INTO SURFACE WATERS;

O. Oenema[1] and C.W.J. Roest[2]

[1] Research Institute for Agrobiology and Soil Fertility (AB-DLO), P.O. Box 14, 6700 AA Wageningen
[2] Winand Staring Centre for Integrated land, Soil and Water Research (SC-DLO), P.O. Box 125, 6700 AC Wageningen

ABSTRACT

The increased input of fertilizers and animal wastes after 1950 has boosted agricultural crop production to a high level in many industrialized countries, but it has also contributed to increased nitrogen and phosphorus emissions from agriculture to groundwaters and surface waters. This paper summarizes the pathways and controls of nitrogen and phosphorus losses to surface waters, and it presents estimates of the losses from agricultural soils in The Netherlands into surface waters. We also present predictions of the effects of policies and measures in The Netherlands' agriculture on nitrogen and phosphorus losses .

KEYWORDS: **agriculture, budgets, nitrogen, phosporus, nutrient losses, controls, policy**

INTRODUCTION

Agriculture is a main source of nitrogen in both groundwaters and surface waters in many parts of the world (e.g. Heathwaite et al., 1993; Kronvang et al., 1995). Agriculture is also an important source of phosphorus (e.g. Foy and Withers, 1995; Rekolainen et al., 1997). The loss of nutrients following the rapid intensification of agriculture from 1950 onwards, has contributed to eutrophication of many surface waters. Since the seventies, a large number of point sources of nitrogen and phosphorus, i.e. fertilizer plant discharges and waste water and urban water discharges, have decreased steadily. Meanwhile, diffuse losses of phosphorus from agriculture remained at the same level or increased (e.g. Uunk, 1991).

The increased input of fertilizers and animal wastes into agricultural land is the main cause for increased losses of nitrogen and phosphorus to surface waters. In The Netherlands, fertilizers and animal wastes make up more than 90% of the total net nitrogen and phosphorus inputs to agricultural soils. Nearly 50% of the total net nitrogen input is harvested again in crops. Of the remainder, some is returned to the atmosphere via denitrification, some is retained in the soil, some enters underground aquifers from where it eventually returns to surface waters and some is directly emitted from the soil to surface waters via runoff and drainage water. Similar data can be presented for other industrialized countries. Evidently, the loss of nitrogen from agriculture is complex and involves a large number of processes and factors (e.g. Burt et al., 1993). Because of the large-scale intensification of agriculture in the European Union, 'the nitrate and phosphorus issues' have shifted in scale from what was once a local pollution problem to what is now one of regional dimensions (Heatwaite et al., 1993).

The purpose of this paper is (i) to indicate the various pathways and controls of agricultural nitrogen and phosphorus discharges into surface waters, (ii) to provide a brief overview of the present nitrogen and phosphorus losses from The Netherlands' agriculture into surface waters, and (iii) to summarize the predicted changes in nitrogen and phosphorus losses to surface waters, following the implementation of policies and measures to reduce nutrient losses from agriculture in The Netherlands.

PATHWAYS OF NITROGEN AND PHOSPHORUS LOSSES FROM AGRICULTURAL LAND INTO SURFACE WATERS

Nitrogen and phosphorus are both essential elements for living organisms, but their biogeochemical functioning in the biosphere is different. Nitrogen is a redox active element, and thereby involved in the transfer of energy, protons and electrons. The mobility of nitrogen compounds alters drastically when nitrogen is biologically transformed from the reduced to the oxidized state and vice versa. All common inorganic nitrogen compounds are highly soluble in water and mobile, causing a rapid dispersion of nitrogen into the wider environment. By contrast, phosphorus has a low solubility and is much less mobile in soils than nitrogen. Transformations and cycling of phosphorus are determined by sorption and precipitation-dissolution reactions. These intrinsic characteristics of nitrogen and phosphorus show up also in different pathways and controls of nitrogen and phosphorus losses from agriculture.

Agricultural systems are leaking nitrogen and phosphorus via three major routes (i) upwards, via crop uptake and via transfer of gaseous nitrogen compounds (NH_3, N_2, N_2O, NO_X) to the atmosphere; (ii) sidewards, via surface runoff, subsurface flow and tile drainage; and (iii) downwards, via leaching. The balance between these different routes depends on the farming system, nitrogen and phosphorus fertilizing history, climate, topography, soil type and hydrology.

Nitrogen losses

Surface waters receive agricultural nitrogen predominantly via:

(1) *Atmospheric deposition.* Approximately 10-25 kg/ha/year of N is deposited by rainfall (wet deposition) in western Europe (Addiscott et al., 1991). About half of this amount is NH_x, and up to 90% of the total NH_x burden in the atmosphere originates from agriculture, with livestock waste as major source (Bussink and Oenema, 1996). Dry deposition and absorption of gaseous nitrogen compounds and aerosols is suggested to be a relatively small nitrogen source for surface waters.

(2) *Subsurface flow and tile drainage.* This pathway is important in imperfectly drained loamy and clayey soils and in soils with shallow groundwater level and/or with a tile drainage system. Studies on arable land show that nitrogen contents in the drains entering surface waters are in the range of 1 to 20 mg per liter. This wide range is a result of differences in nitrogen input on the agricultural land and in nitrogen losses via other pathways, notably denitrification. A special phenomenon in structured soils is by-pass flow or preferential flow associated with macroporosity. By-pass flow is rapid. It may carry fertilizer and animal waste nitrogen following its application directly from the soil surface to the tiles. Alternatively, by-pass flow may tend to be low in solutes, because of the limited interaction with the soil mass. Evidently, flow-proportional sampling is required to obtain an unbiased estimate of the nitrogen discharges from these soils into the surface waters. A two-dimensional approach has to be used to describe water flow and nitrate transport to a subsurface drain. A typical example of Darcian water flow in a layered silt loam soil is shown in Figure 1 (after De Vos, 1997).

(3) *Surface runoff and erosion.* This pathway is important in hillslope areas and undulating land in wet climate. Land use, topography and rainfall distributions are decisive factors. Episodic storm events may contribute up to 90% of the total nitrogen loss via runoff and erosion. On slightly undulating agricultural land on poorly drained clayey soils, with tiny drainage trenches, surface runoff can also be a significant pathway of surface water loading. The loss of nitrogen via surface runoff and erosion is very difficult to quantify, because of the heterogeneous spatial and temporal patterns. In flat areas, such as The Netherlands, surface runoff and erosion are suggested to be a relatively minor pathway of surface water enrichment with nitrogen.

(4) *Animal excreta discharges at watering sites.* Grazing animals may drop urine and dung in the surface water when drinking. This route of surface water enrichment has been drastically decreased in many areas following fencing and the construction of special drinking-water facilities for grazing animals.

(5) *Seepage of nitrate containing groundwater from agricultural land.* Seepage occurs on the foot of hillslopes. The outlet seepage water may have entered the aquifers via leaching through agricultural soils far ahead, and a number of transformations processes may have altered the composition of the outlet water. In the polder areas of The Netherlands, high ammonium and phosphorus concentrations are found in the seepage water (RIVM, 1992). The nitrogen and phosphorus originate from the mineralization of organic-rich marine sediments.

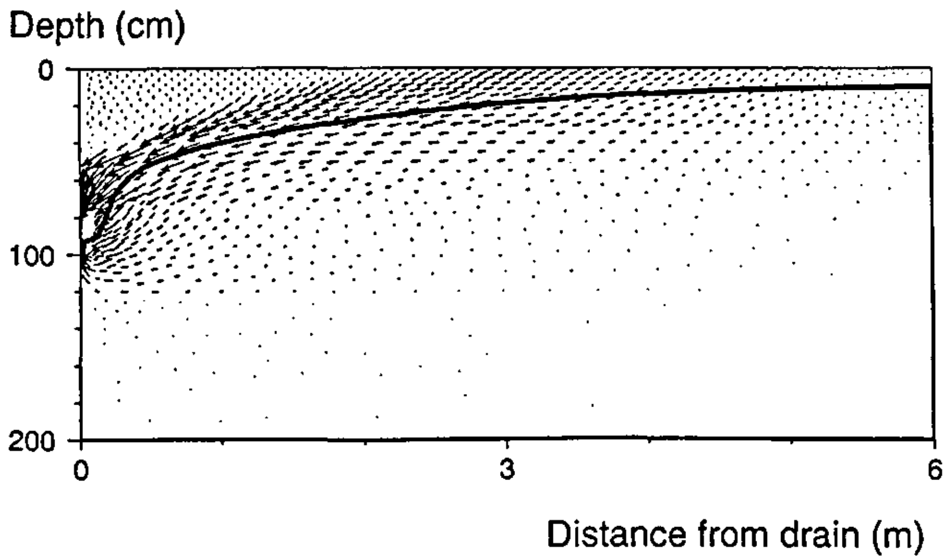


Fig. 1. Simulated water flux densities in the soil profile under wet conditions. The arrows indicate the direction and magnitude of the water flux density (maximum water flux density = 50 cm/d). The water table is indicated by the drawn line and the saturated zone (groundwater) is dashed. Because there is a plane of symmetry midway between the drains, the results are only presented for half the drain spacing of 6 m. At a depth of 2 m the soil is assumed to be impermeable (De Vos, 1997).

Phosphorus losses

Interest in the role of phosphorus from agriculture in surface water eutrophication has emerged only recently. Excellent recent reviews on the movement of phosphorus from agriculture to surface waters (e.g. Uunk, 1991; Sharpley and Rekolainen, 1997; Sharpley and Withers, 1994; Sharpley et al., 1994; Rekolainen et al., 1997; Pionke et al., 1997; Chardon et al., 1996), have clearly indicated that agriculture is an important diffuse source of phosphorus, and that phosphorus can be transported from agricultural land to water bodies by various pathways. The widely held belief that soils strongly retain phosphorus and that surface water enrichment with phosphorus is unlikely to be coming from agricultural soils, cannot longer be upheld. This view has emerged concomitant with the observation that eutrophication of surface waters have remained following the clean-up of major point sources of phosphorus from the early seventies onwards. Attention is now being directed more and more towards agriculture as a diffuse source of phosphorus. It should be noted that the loss of phosphorus from agriculture is not a great economic loss for the farmer, but the ecological consequences of the phosphorus in the surface waters can be large (e.g. Uunk, 1991).

Unlike nitrogen, the major proportion (60-90%; Sharpley and Rekolainen, 1997) of phosphorus transported from agricultural soils is in particulate form, generally. Also unlike nitrogen, only a fraction of the phosphorus transported from agricultural land is directly bioavailable to aquatic biota. The bioavailability of particulate phosphorus can vary from 10 to 90%. Dissolved organic phosphorus may constitute up to 60% of the total dissolved phosphorus in soils and up to 10% of the total P load in surface runoff (Chardon et al., 1997). The bioavailability of dissolved organic phosphorus appears to depend heavily on the conditions in the surface water, which govern the transformation of organic into inorganic phosphorus. The bioavailable particulate phosphorus and total dissolved phosphorus (the major part) constitute the bioavailable phosphorus.

There are two major pathways of phosphorus transport from agricultural land to surface waters:
(a) *Surface runoff and erosion.* Soil erosion and surface runoff are important in hilly areas, where water-infiltration rate is low. Such losses are strongly related to storm events and prolonged rainfall, and can be increased further where animal waste and phosphorus fertilizer have been applied just prior to such an event. A number of studies have shown that the loss is proportional to the phosphorus status of the top soil; the higher the phosphorus status the higher the discharges of agricultural phosphorus into surface waters (e.g. Sharpley

and Rekolainen, 1997). Overall, the movement of phosphorus to water bodies via surface runoff and erosion is a discontinuous process, driven mainly by rainfall distribution, hydrology and farm management.

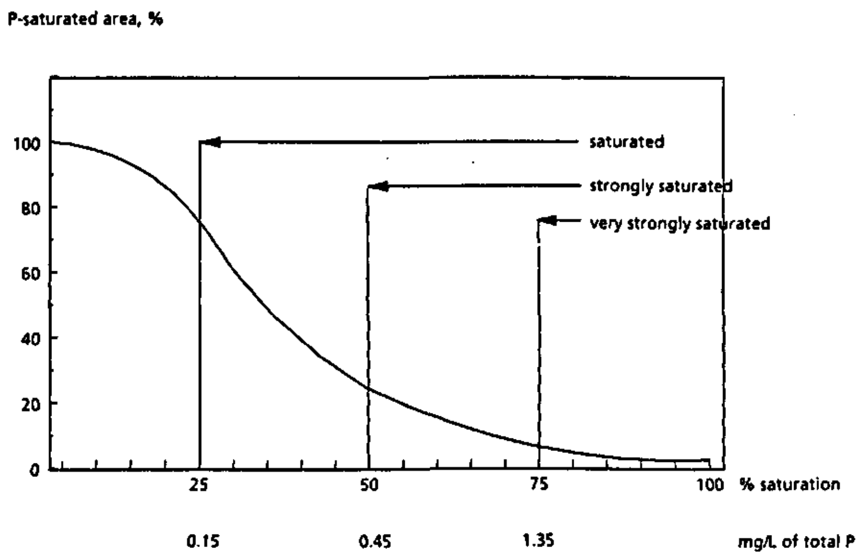


Fig. 2. Relationship between the percentage saturation of the phosphorus adsorption capacity of the soil and the percentage of the total agricultural area on sandy soils that can be identified as phosphate leaking. (after Reijerink and Breeuwsma, 1992). A phosphorus saturation of the soil of 25, 50 and 75% yield a concentration of total dissolved phosphorus in the upper groundwater of 0.15, 0.45 and 1.35 mg per liter, respectively. A total of 78% of the agricultural area on sandy soils had a phosphorus saturation percentage of at least 25% in 1992.

(b) *Subsurface flow and tile drainage.* In well-drained agricultural soils, the phosphorus concentration in the deep groundwater is generally less than 0.01 mg/L (Van Swinderen et al., 1996). In poorly drained, sandy soils heavily loaded with phosphorus containing animal wastes and fertilizers, phosphorus concentrations in shallow groundwater can be much higher (Breeuwsma and Schoumans, 1986; Chardon et al., 1996), because the capacity of these soils to adsorb phosphorus from the percolating soil solution is small. Once the phosphorus adsorption front reaches the groundwater level, its discharge into surface water greatly accelerates. Such soils may discharge more than 100 times the amount of phosphorus discharged by well-drained soils with a phosphorus-poor subsoil. Consequently, these soils act as hot spots, as point sources of surface water enrichment. It has been indicated that the phosphorus concentrations in the upper groundwater of sandy, acidic soils exceeds a level of 0.15 mg/L when more than 25% of the total phosphorus adsorption capacity of the soil has been utilized (Van der Zee et al., 1990). Concentrations of phosphorus in the groundwater progressively increase when the percentage phosphorus saturation of the soil increases. Therefore, priority should be given to identify and remediate these soils. The total surface area of The Netherlands covered with phosphorus leaking sandy soils has been estimated to range between 300,000 and 400,000 ha (Reijerink and Breeuwsma, 1992), depending on the criteria used for leaking soils (Fig. 2).

CONTROLS OF NITROGEN AND PHOSPHORUS LOSSES FROM AGRICULTURAL LAND INTO SURFACE WATERS

A large number of factors control nitrogen and phosphorus losses from agricultural land into surface waters. These controls can be subdivided into natural controls and farm management controls. Natural controls are dynamic and highly variable in both space and time. The most important natural controls are:
- climate;
- topography and hydrology; and
- soil characteristics.

These three factors determine both potential crop growth, potential nitrogen and phosphorus losses, and the partitioning of nitrogen losses between leaching, runoff, denitrification and subsurface flow. The impact of these controls on nitrogen have been studied extensively during the last couple of decades, and first management plans and schemes (see below) have been set up already on the basis of an understanding of the impacts of these natural controls on nitrogen losses. The natural controls of phosphorus losses also have been studied extensively during recent decades, but the hydrological controls linking spatially variable phosphorus sources, sinks and transport processes within the landscape are still less well understood (Sharpley and Rekolainen, 1997). This hampers the development of effective management programmes addressing the decrease of phosphorus losses into surface waters. It has been suggested that up to 90% of the annual phosphorus losses occurs from only 5% of the land during only one or two storms, especially in areas where surface runoff and erosion are the dominant routes for phosphorus losses. Linkage of run-off producing variable source areas and areas of high soil phosphorus status is essential to delineate critical-source-area controls of phosphorus losses and to indicate where remediation efforts may be best directed (e.g. Pionke et al., 1997). Similar conclusions hold for phosphorus leaking soils, because of the uneven spatial distribution of phosphorus in these phosphorus rich-soils and the large temporal and spatial changes in groundwater level and flow patterns. Identification of these hot spots in the landscape is difficult but an essential step for the effective control of losses from these soils (Chardon et al., 1996).

The natural controls may result in completely out-of phase discharges of nitrogen and phosphorus into water bodies via e.g. tile drainage (Fig. 3). Nitrate concentrations in the drainage water are controlled by the amount of nitrate in the soil and by climate, i.e. precipitation excess. Results of a seven-years lasting lysimeter experiment show that nitrogen applications via animal waste and fertilizer in spring were larger than nitrogen removal via the crop (Zea mays L), and that the excess nitrate was leached during autumn and winter. Peaks of nitrate in the collected drainage water occurred at the end of the year when the excess precipitation had washed the residual mineral nitrogen down to the 40 cm thick soil layer. Nitrate concentrations were low in spring, because essentially all nitrate had been washed out of the soil by that time. By contrast, peaks of dissolved total phosphorus occurred in spring, when nitrate concentrations were low. Low phosphorus concentrations coincided with high nitrate concentrations. This contrasting behaviour of nitrate and dissolved phosphorus has been attributed to different controls. Sorption of phosphorus is depending on among other ionic strength of the soil solution, and the coincidence of high phosphorus concentrations with low nitrate and chloride concentrations is indicative for the importance of the chemical control of phosphorus leaching (Chardon et al., 1997). It has to be proven yet whether this has consequences for the nitrogen and phosphorus availability to aquatic organisms.

Farm management can strongly modify the impact of the natural controls on nitrogen and phosphorus losses. Most important farm management controls are:
- land use and crop rotation;
- animal waste and fertilizer applications, both amount and method of application;
- grazing system and management;
- soil phosphorus management and erosion control;
- groundwater level control (drainage); and
- irrigation management.

Effective nutrient management planning and effective execution of best management practices requires a thorough understanding of the behaviour of nitrogen and phosporus in the system and of the controls of the losses. This understanding is still incomplete. Moreover, the perceived economic costs and risks of strict nutrient management planning may make farmers reluctant to adopt some of the measures, without further incentives. However, a number of governmental policies, obligatory measures and legislations have given farmers the impetus to improve the management and to decrease the losses.

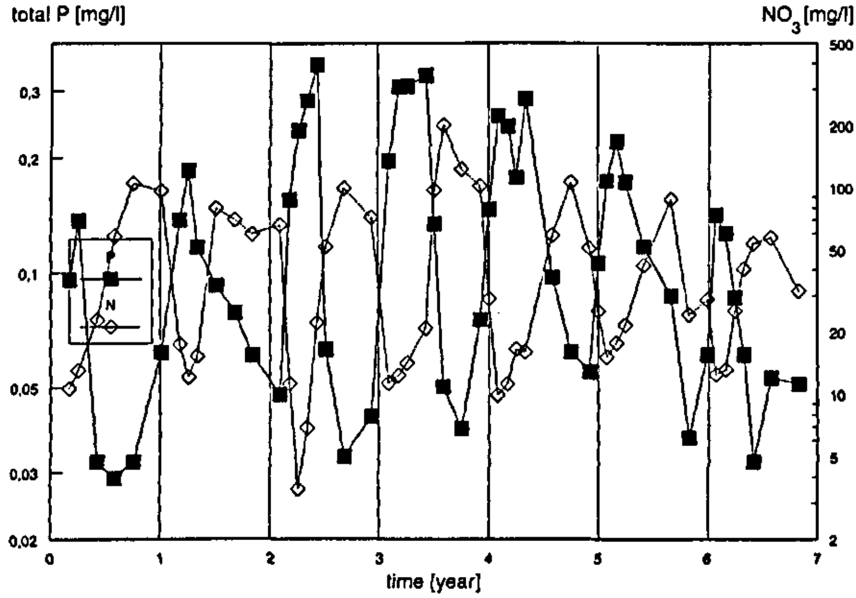


Fig. 3. Monthly averages of nitrate and total dissolved phosphorus concentrations in the leachates of lysimeters. The outdoor lysimeters contained 40 cm of sandy soil (2% clay and 4% organic matter), and received heavy applications of nitrogen fertilizer and animal wastes each spring. The soil was cropped with Zea mays. Leachates were sampled and analyzed weekly for seven years (Chardon et al., 1997).

Effective policies and measures encroach on the strategic and tactical goals and decisions of the farm management. Measures that require decisions at the operational level are effective only when embedded in the farm management strategy. Hence, best management practices must also have an embedding in the strategic goals of the farm management. Best management practices also require a tailor-made design at farm, plot and even subplot level, and should be designed as a means to reach the strategic goals and boundary conditions.

Best management practices must be based on a sound analyses of nutrient cycling and losses, and of the controlling factors. Best management practices include strategies to prevent losses. Proper strategies should focus on (e.g. Sharpley and Rekolainen, 1997):

(a) *Source management measures*; i.e. minimizing off-farm inputs and surpluses of nitrogen and phosphorus, effective utilization of nitrogen and phosphorus from animal wastes, crop residues, and soil. Evidently, soil tests should receive much greater emphasis in nutrient management planning than it has received on many intensive livestock farms in the recent past. Frequency distribution of soil phosphorus test results of grassland and land used for fodder maize show a large percentage of fields that have a high phosphorus status (Oenema and Van Dijk, 1994). Evidently, these soils carry a high risk for loading surface water with too large amounts of phosphorus via surface runoff and leaching. Application of phosphorus containing fertilizers and animal wastes on such fields is of no use for plant growth and may seriously increase the risk of phosphorus loss.

(b) *Transport management measures*; control of soil erosion and runoff by vegetative cover, buffer strips, terracing, etc. (Kruijne, 1996). Manipulating groundwater levels may greatly affect total nitrogen and phosphorus losses and also the partioning of the various pathways. However, the control of groundwater level is complex and interferes also with soil cultivation ability. Often, the effects on nitrogen and phosphorus are opposite. A shallow groundwater table increases the potential for denitrification and thereby decreases nitrate leaching losses, but greatly increases the risk of phosphorus leaching to the surface waters.

The grazing system is an important management instrument as regards to the partioning of nitrogen losses via nitrate leaching and ammonia volatilization (e.g. Van der Meer and Van der Putten, 1995).
(c) *Remedial management measures* When noting that 90% of the annual phosphorus load occurs from 5% of the land during only one or two storm events, it is evident that identifying these hot spots is an essential first step. Second step is the remediation of these hot spots. Soils with a high phosphorus status and/or a high index for phosphorus saturation, should be subjected to a remediation management strategy.
Evidently, no precise guidelines can be prescribed for all farms. Each farm should adopt a series of best management practices to reach the environmental and economic objectives at strategic management level.

EMISSIONS OF NITROGEN AND PHOSPHORUS FROM THE NETHERLANDS' AGRICULTURE

Until 1987, total input of nitrogen and phosphorus in agricultural land via fertilizers and animal wastes steadily increased, especially in livestock farming systems with intensively managed grassland and maize land (CBS, 1992). Following a number of policy measures, e.g. implementation of milk quota, a ban on the further intensification of intensive livestock farming, and more advise to farmers, total nitrogen and phosporus input decreased between 1987 and 1990. Results presented in Table 1 for the year 1994 indicate that inputs of nitrogen and phosphorus by far exceed the output via crop removal in The Netherlands' agriculture .

Agriculture is the major emitter of nitrogen to surface waters (Figure 4). Whilst the total loading of surface waters with nitrogen and phosphorus has diminished between 1986 and 1995, especially for phosphorus, the contribution of agriculture remained at the same level, and thus increased percentage-wise (CBS, 1997), despite the overall decrease in N input in agriculture. There is, however, a delayed-response, because significant decreases in nitrogen emissions from agricultural land are expected indeed for the years to come (see next sections).

Table 1. Mean nitrogen and phosphorus budgets of the agricultural land in The Netherlands in 1994 in Gg per year. Inputs via animal wastes and fertilizers are corrected for losses via ammonia volatilization and represent total net input of nitrogen. Fong, 1997, CBS, 1992, 1997.

Description of items	Nitrogen	Phosphorus
Inputs		
Fertilizers	365	30
Animal wastes (net)	509	92
Atm. deposition	68	2
Other	38	5
Total net input	980	129
Output		
Harvested crops	447	56
Input - Output		
Net loading of the soil	533	73
Ibid, in kg per ha per year	267	37

There are significant regional differences in total nitrogen input in agriculture and in the fate of the excess nitrogen. On the clay and peat soils in the North, West and Central Netherlands where arable farming and specialized dairy farming dominate, farming is less intensive and total nitrogen input is smaller than on the sandy soils in the East and South Netherlands where highly intensive dairy farming and pig breeding dominate.

These two regions also differ in hydrology and groundwater levels, and thereby also in nutrient transformations and losses (Roest & Boogaard, 1997) . The sandy soils in the East and South, covering 37% of the agricultural area, are situated above mean sea level, and are free draining to the underground water resources. As a result, nitrate concentrations in the shallow groundwater of these sandy soils exceed on average the 1980 EC Directive on the Quality of Water intended for human consumption (Van Swinderen et al., 1996). By contrast, the clay

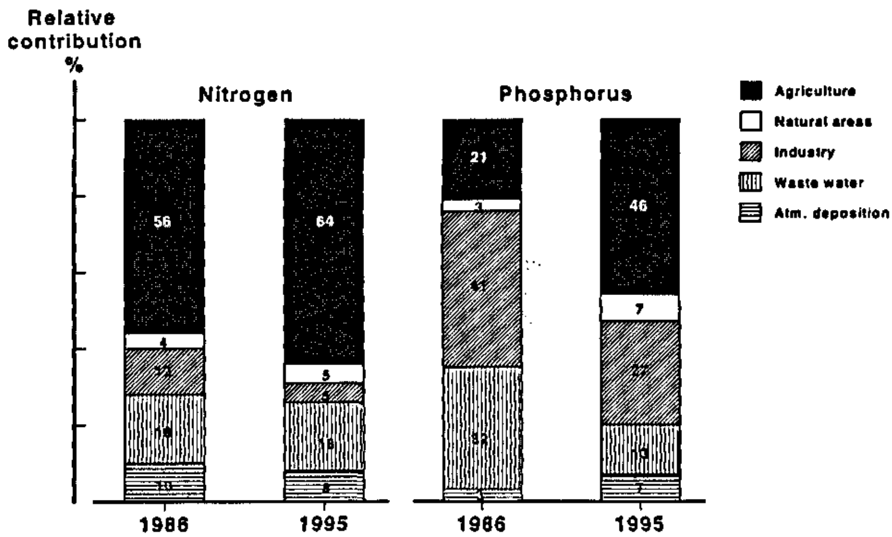


Figure 4. Contribution (in %) of various sectors to the total nitrogen and phosphorus loadings of fresh surface waters (lakes, rivers, canals) in The Netherlands. Total nitrogen loadings were 186 and 168 Gg, and total phoshorus 34 and 15 Gg in 1986 and 1995, respectively (CBS, 1997).

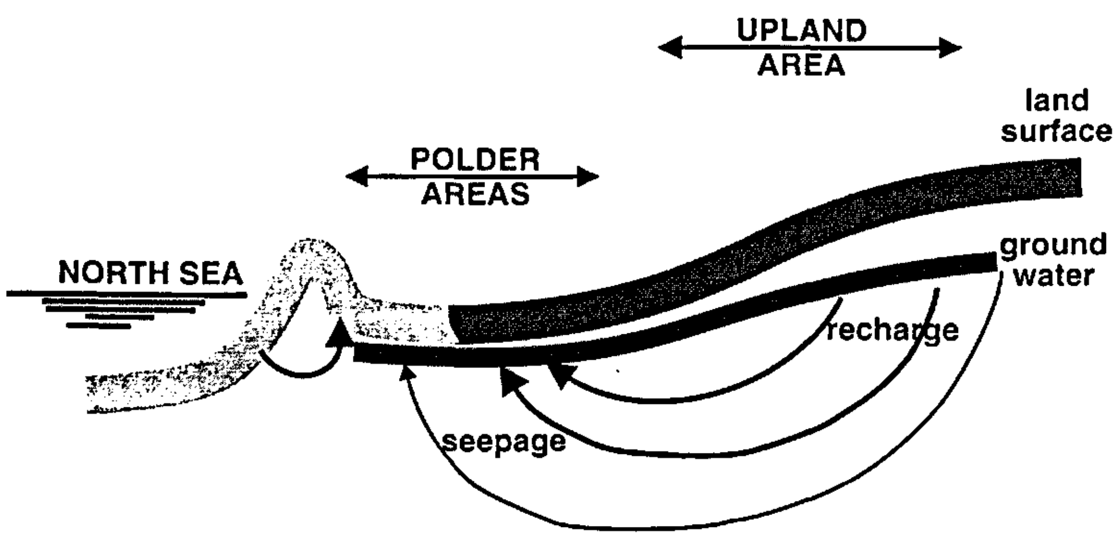


Figure 5. Simplified West-East cross section of The Netherlands showing the groundwater recharge areas on sandy soils in the East and South, and the polders influenced by seepage in the North and West.

and peat soils, covering 47 and 14% of the agricultural area, respectively, are situated below mean sea level, have an artificial drainage system and are influenced by seepage (Figure 5). Nitrate concentrations in the groundwaters of clay soils are relatively low; on average less than 50 mg per liter, but on about 20% of the drained clay soils nitrate concentration exceed 50 mg per liter. Nitrate concentrations in the groundwaters of peat soils are low, because denitrification is a major pathway of nitrogen losses in these soils (Huinink & De Waard, 1997).

NITROGEN AND PHOSPHORUS EMISSIONS FROM AGRICULTURE IN THE NETHERLANDS FOLLOWING THE IMPLEMENTATION OF MINAS

The seriousness and scale of the nitrogen contamination of the environment and the phosphorus loading of agricultural soils have been the stimulus of a series of policies and measures launched by the Netherlands' government from 1985 onwards. A step-wise tightening of policies and measures was envisaged, to provide farmers the opportunity to adapt to a more efficient nutrient management. The first measure was a ban on the further intensification of intensive livestock farms. The last step in this series of policies and measures is the implementation of MINAS, a nitrogen and phosphorus accounting system at farm level, in combination with criteria for levy-free surpluses of nitrogen and phosphorus. Between 1998 and 2008, levy-free surpluses gradually decrease (Anonymous, 1995). These final policy measures aim at a 50 % lowering of all nitrogen and phosphorus emissions from agriculture, including the emissions into surface waters, with reference to 1985. Additional policies and measures will be implemented regional wise, because MINAS neglects differences between soils, landscape and catchments in the control of nitrogen and phosphorus losses. However, these additional policies and measures still need to be operationally defined.

To predict the effects of the implementation of MINAS, the Netherlands was divided into 3624 units, each having characteristic land use type, soil type and hydrology. Net nitrogen and phosphorus surpluses at farm level were based on the assumption that farmers will meet the levy-free surpluses. The soil and hydrological conditions reflect mean actual conditions, using data from the national soil data base. Computations were made with sequences of a 15 year meteorological data base. The modeling tools have been calibrated and validated in various studies. For a more detailed description, the reader is referred to Roest & Boogaard (1997) and Boers et al., (1996).

Predictions of the overall mean nitrate concentrations in the groundwater of all agricultural land are presented in Table 2. The year 2015 has been chosen as the reference year for the full implementation of MINAS, when essentially all possible delayed-response effects have faded away. Following the implementation of MINAS, mean nitrate concentration at 1 meter below the average lowest groundwater level (ALG-1) will decrease to an average of about 11 mg per liter in the year 2015, and the agricultural area with a nitrate concentration in the groundwater at ALG-1 that exceeds 50 mg per liter will decrease to about 7%. The latter area is mainly confined to dry sandy soils.

There is a considerable noise in the predicted mean nitrate concentrations and in the predicted area with too high nitrate concentrations, which is associated with the reference depth of the groundwater level. Nitrate concentrations are higher at the average lowest groundwater level (ALG) than at 1 meter below the average lowest groundwater level (ALG-1), because of differences in residence time of the groundwater (about 12 months) and, hence, in nitrate removal through denitrification. As a result, the area with too high nitrate concentrations in the year 2015 is about twice as large with ALG as reference depth than with ALG-1 as reference depth (Table 2). Approximately 20% of the agricultural area will have a nitrate concentration at ALG that exceeds 50 mg per liter. At ALG-1, this area has reduced to 7%.

Table 2. Predicted mean nitrate concentrations in the groundwater of agricultural soils at two depths, following the implementation of the nitrogen accounting system MINAS in 1998. Mean nitrogen surplus has been calculated following the surface balance approach. Also presented is the percentage agricultural area where nitrate concentrations in the groundwater will exceed 50 mg per liter (Roest & Boogaard, 1997).

Year	Nitrogen surplus in kg ha^{-1} yr^{-1}	Nitrate concentrations in mg NO_3 per liter at ALG	at ALG-1	Area exceeding 50 mg NO_3/l in % at ALG	at ALG-1
1985	385	51	28	22	17
1995	310	58	35	35	17
1998	248	55	34	30	20
2000	223	53	33	29	20
2002	198	48	28	27	17
2005	162	38	21	25	16
2008	147	35	18	23	16
2015	147	25	11	21	7

Predicted changes in the total nitrogen and phosphorus surplus and in the emissions from agricultural land to surface waters between the reference year 1985 and the year 2008 have been summarized in Table 3. The mean nitrogen surplus at farm level decreases with more than 60% between 1985 and 2008. However, the predicted decrease in the nitrogen emissions to surface waters is much smaller than 60%. The relatively small decrease in the total emission of nitrogen form agriculture to surface waters between 1985 and 2008 is partially due to a delayed response reaction, partially due to changes in the mineralization - immobilization equilibrium of organic nitrogen in agricultural soils. Note that the decrease between 1998 and 2008 in the emissions of nitrogen to surface waters is 57%. Evidently, the nitrogen emissions to surface waters will decrease significantly, albeit not with 50% between the years 1985 and 1995.

Table 3. Predicted mean nitrogen and phosphorus emissions to surface waters as a function of mean nitrogen and phosphorus surplus at farm level, calculated according to the MINAS guidelines, following the implementation of the nutrient accounting system MINAS between 1998 and 2008. Data in brackets indicate the relative changes compared to the reference year 1985 (in per cent). (WSV, 1996)

Year	Nitrogen surplus (kg ha^{-1} yr^{-1})	Nitrogen emissions to surface waters (kg ha^{-1} yr^{-1})	Phosphorus surplus (kg ha^{-1} yr^{-1})	Phosphorus emission to surface water (kg ha^{-1} yr^{-1})
1985	385	29*	49	1.7**
1995	310 (-19)	35 (+20)	31 (-36)	1.8 (+8)
1998	248 (-36)	36 (+24)	17 (-65)	1.8 (+7)
2000	223 (-42)	33 (+15)	15 (-69)	1.8 (+6)
2002	198 (-49)	31 (+ 5)	13 (-73)	1.8 (+5)
2005	162 (-58)	23 (- 21)	11 (-78)	1.8 (+4)
2008	147 (-62)	19 (- 33)	9 (-82)	1.7 (+4)

* excluding 5 kg/ha/yr from upward seepage
** excluding 1,23 kg/ha/yr from upward seepage

Total phosphorus surplus in agriculture will have decreased to about 20 Mg by the year 2008 i.e. a mean of about 8 kg/ha/year. Total loading of soils with phosphorus will decrease by more than a factor 4. This decrease requires a huge effort of the agricultural sector (Boon-Prins et al., 1996). Despite the huge decrease in phosphorus input in The Netherlands' agriculture, losses of phosphorus to surface water are expected to decrease only slightly (table 3). The predicted emission of phosphate to the surface waters still increased until 1995. The small decrease after 1995 is due to banning of surface application of animal slurry in The Netherlands. Simultaneously, drainage discharges will increase (table 4) due to the continuing loading of the soil system with phosphate.

Table 4. Calculated average nitrogen and phosphorus balances and partitioning of losses for The Netherlands' agriculture (soil system till 7 m depth) in 1985 (history), 1995 (present) and 2008 (future; after MINAS implementation), in kg/ha/yr.

Item	1985	1995	2008
NITROGEN			
MINAS surplus	385	310	147
Deposition	29	28	19
Upwards seepage	5	5	5
Total soil input	419	343	171
Volatilization	44	27	13
Denitrification	185	193	153
Runoff	2	2	1
Drainage	32	38	25
Recharge	14	19	16
Total soil losses	277	279	208
Soil storage	+142	+ 64	- 39
PHOSPHORUS			
MINAS surplus	49	31.4	8.7
Deposition	0.6	0.6	0.6
Upward seepage	0.5	0.5	0.5
Total soil input	50.1	32.6	9.9
Runoff	0.2	0.2	0.1
Drainage	2.1	2.2	2.1
Recharge	0.1	0.1	0.1
Total soil losses	2.3	2.5	2.4
Soil storage	+48.2	+ 30.1	+ 7.5

The soil system is known to be a large reservoir for organic and inorganic nitrogen and phosphorus. Between 1995 and 2008 the introduction of the MINAS accounting system change the nitrogen balance of the soil from an increase to a decrease in soil nitrogen (table 4). Depletion of soil nitrogen in the year 2008, after full implementation of MINAS indicates that soil losses are still decreasing in time. Consequently, lower surface water loads may be expected during the years beyond 2008.

Although the rate of loading of soils with phosphate will have been reduced by about 85% in the year 2008 (table 3), the present criteria in the MINAS accounting system are clearly insufficient to reverse the trend of loading soils with phosphorus (table 4). A further increase in surface water emissions with phosphate can thus be expected beyond the year 2008, unless additional measures are taken. Additional policies and measures are being prepared now for phosphorus leaking soils only (e.g. Fraters and Boumans, 1997).

Summarizing, implementation of the nutrient accounting system MINAS, with a step-wise lowering of nitrogen and phosphorus surpluses at farm level, will decrease the total nitrogen losses between the years 1985 and 2008 with about 60%. This large decrease is the result of less input via fertilizers and animal wastes, combined with a similar crop offtake. Nitrogen emissions from agricultural land to surface waters also decrease significantly, albeit less than 50% between 1985 and 2008. Phosphorus surplus will decrease with more than 80%, but phosphorus losses to surface waters are expected to decrease only slightly.

REFERENCES

Addiscott, T.M., Whitmore, A.P. & Powlson, D.S. (1991): Farming, Fertilizers and the Nitrate Problem, CAB International, Wallingford.

Anonymous (1995). Integrale Notitie mest- en ammoniakbeleid. Ministerie van Landbouw, Natuurbeheer en Visserij, SDU Uitgeverij, 's Gravenhage, 39 pp.

Anonymous (1996). Mest- en ammoniakbeleid. Nieuwsbrief, Ministeries LNV en VROM, 12 pp.

Boers, P.C.M., Boogaard, H.L., Hoogeveen, J., Kroes, J.G., Noij, I.G.A.M., Roest, C.W.J., Ruijgh, E.F.W. & Vermulst, J.A.P.H. (1997). Huidige en toekomstige belasting van het oppervlaktewater met stikstof en fosfaat vanuit de landbouw. RIZA rapport 97.013 & SC-DLO rapport 532 (in press).

Boons-Prins, E.R., Van der Meer, H.G., Sanders, J., Tamminga, S., Van Vugt, F. & De Wilt, J.G. (1996). Drastische verbetering van de nutriëntenbenutting in de dierlijke produktie. NRLO, Den Haag, NRLO-rapport nr. 94/3, 55 pp., 14 bijl.

Breeuwsma, A. & Schoumans, O.F. (1986) Fosfaatophoping en -uitspoeling in de bodem van mestoverschotgebieden. Bodembeschermingsreeks, nummer 74. Staatsuitgeverij 's Gravenhage

Burt, T.P., Heathwaite, A.L. & Trudgill, S.T. (Eds.) (1993). Nitrate: Processes, Patterns and Management. John Wiley & Sons Ltd, Chichester, 444 pp.

Bussink, D.W. and O. Oenema (1997). Ammonia volatilization from dairy farming systems in temperate areas; a review. Nutr. Cycl. Agroecosyst. (in press).

CBS (1992). Mineralen in de landbouw 1970-1990. Staatsuitgeverij, Den Haag.

CBS (1997). Boeren in een veranderend milieu. (In preparation).

Chardon, W.J., Oenema, O., Schoumans, O.F., Boers, P.C.M., Fraters, B. & Geelen, Y.C.W.M. (1996). Verkenning van de mogelijkheden voor beheer en herstel van fosfaatlekkende landbouwgronden. Programma Geïntegreerd Bodemonderzoek, Wageningen. Rapporten Programma Geïntegreerd Bodemonderzoek; deel 8, 55 p., 6 bijl.

Chardon, W.J., Oenema, O., Del Castilho, P., Vriesema, R., Japenga, J. & Blaauw, D. (1997). Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environm. Qual. 26: 372-378.

De Vos, J.A. (1997). Water flow and nutrient transport in a layered silt loam soil. PhD-thesis, Wageningen Agricultural University (in press).

Fong, N. (1997). Mineralen in de landbouw (1995). Kwartaalbericht Milieustatistieken no. 3.

Foy, R.H. & Withers, P.J.A. (1995). The contribution of agricultural phosphorus to eutrophication. The Fertiliser Society, Proceedings no. 365, 32 pp.

Fraters, B. and L.J.M. Boumans (1997) Technical background of measures to remediate phosphorus leaky soils (in Dutch). RIVM nr. 716601001, Bilthoven (in press).

Heathwaite, A.L., Burt, T.P. & Trudgill, S.T. (1993). Overview - the Nitrate Issue. In: T.P. Burt, A.L. Heathwaite & S.T. Trudgill (Eds.) Nitrate: Processes, Patterns and Management. John Wiley & Sons Ltd, Chichester, p. 3-21.

Huinink and De Waard. (1997). Drainwater-monitoringproject Hoeksche Waard. IKC rapport, Ede, 87 pp.

Kronvang, B., Grant, R., Larsen, S.E., Svendsen, L.M. & Kristensen, P. (1995). Non-point-source Nutrient Losses to the Aquatic Environment in Denmark: Impact of Agriculture. Mar. Freshwater Res. 46: 167-177.

Kruijne, R. (1996). Perceelsonderzoek naar de effecten van beekbegeleidende bufferstroken op de stikstof- en fosforbelasting van de Mosbeek. Wageningen, DLO-Staring Centrum Rapport 420.2.

Meer, H.G. van der & Van der Putten, A.H.J. (1995). Reduction of nutrient emissions from ruminant livestock farms. In: G.E. Pollott (Ed.) Grassland into the 21st Century: Challenges and Opportunities. Proceedings of the 50th Anniversary Meeting of the British Grassland Society, Harrogate, 4-6 December, 1995, p. 118-134.

Oenema, O. & Van Dijk, T.A.(Eds.) (1994). Fosfaatverliezen en fosfaatoverschotten in de Nederlandse landbouw. Rapport van de technische projectgroep 'P-deskstudie'. Project Verliesnormen, Deelrapport I, pp.102.

Pionke, H.B., Gburek, W.J., Sharpley, A.N., and Zollweg, J.A.. (1997) Hydrological and chemical controls on phosphorus loss from catchments. In: Tunney,H., Carton,O.T., Brookes,P.C. and Johnston,A.E. (eds). Phosphorus loss from soil to water. CAB International 1997, pp. 225-242.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. (1994) Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437-451.

Reijerink, J.G.A. & Breeuwsma, A. (1992) Ruimtelijk beeld van de fosfaatverzadiging in mestoverschotgebieden. Staring Centrum Wageningen, Rapport no. 222.

Rekolainen, S. Ekholm, P., Ulén, B. & Gustafson, A. (1997). Phosphorus losses from agriculture to surface waters in the Nordic Countries. In: Tunney, Carton, O.T., Brookes, P.C. & Johnston, A.E. (eds.), Phosphorus loss from soil to water. CAB International, Wallingford, p. 77-93.

RIVM (1992). Milieudiagnose 1991.III, Bodem en grondwaterkwaliteit. Bilthoven, RIVM, Laboratorium voor Bodem- en Grondwaterkwaliteit.

Roest, C.W.J. & Boogaard, H.L. (1997). Results present and future surface water loads with nitrogen and nitrate concentration in groundwater computed with the WSV model instrument. DLO-Winand Staring Centre, Wageningen, 14 pp.

Sharpley, A.N. & S. Rekolainen (1997). Phosphorus in agriculture and its environmental implications. In: Tunney, H., Carton, O.T., Brookes, P.C. & Johnston, A.E. (eds.), Phosphorus loss from soil to water. CAB International, Wallingford, p. 1-53.

Sharpley, A.N. & Withers, P.J.A. (1994). The environmentally-sound management of agricultural phosphorus. Fertilizer Res. 39: 133-146.

Swinderen, E.C. van, Fraters, B., Vissenberg, H.A., De Haan, T. & De Hoop, D.W. (1996). Meetprogramma: Kwaliteit Bovenste Grondwater Landbouwbedrijven; resultaten tweede bemonstering 1993. RIVM-rapport 714831001, 90 pp.

Uunk, E.J.B. (1991). Eutrophication of surface waters and the contribution of agriculture. The Fertiliser Society, Proceedings no. 303, 55 pp.

Zee, S.E.A.T.M. van der, Van Riemsdijk, W.H. & De Haan, F.A.M. (1990) Het protokol fosfaatverzadigde gronden. Deel I: Toelichting. Wageningen, Landbouwuniversiteit, Vakgroep Bodemkunde en Plantevoeding.

WSV (1996). Landelijke Watersysteemrapportage 1994.

ASSESSMENT OF NITROGEN INPUTS FROM AGRICULTURAL AND URBAN NON-POINT SOURCES: THE GAZA STRIP (CASE STUDY)

Habib Muhammetoglu (Mukhallalati)

Akdeniz University, School of Technology (Teknik Bilimler M.Y.O.), 07059 Antalya, Turkey.
E-Mail: Habib @ ufuk.lab.akdeniz.edu.tr

Cors van den Brink

IWACO Consultants for Water and Environment, P.O.Box 8520, 3009 AM Rotterdam, The Netherlands. E-Mail : krdw@noord.bart.nl

ABSTRACT

In developing countries, eutrophication of surface and subsurface waters may arise from the following:

(i) Absence of wastewater collection and treatment facilities

(ii) Excess use of fertilizers and irrigation water in agriculture

(ii) Uncontrolled solid waste dumping areas

The Gaza Strip is a good example for such a case. The Gaza Strip is a small coastal area along the eastern Mediterranean Sea, densely populated with more than one million inhabitants. The main source of water supply in the Gaza Strip is groundwater. Nitrate is a well-documented pollutant of drinking water in the Gaza Strip. Typical nitrate concentrations are in the range of 100-200 mg NO_3 /l, with extreme values up to 650 mg NO_3/l. Main causes of nitrate pollution are believed to be leachate of wastewater and solid waste dumps in the urban areas and excess use of fertilizers and irrigation water in the rural areas. With respect to wastewater, the collection system covers only a small part of the Gaza Strip. The other part depends on percolating vaults or cesspits to collect wastewater which afterwards leaches to groundwater. With respect to irrigation return flow, excessive fertilizer application and irrigation water is a common practice in agriculture especially for the green houses, tunnels and horticulture. With respect to solid waste dumps, nitrate-rich leachate originating from all the landfills is allowed to pass to groundwater. A recent study has been carried out to assess the nitrogen balance for both the agricultural rural areas and the urban areas taking into consideration the nitrogen application in fertilizers, organic manure, nitrate-rich irrigation water and wastewater, and processes such as volatilization, nitrification, denitrification, and plant uptake. The results show that nitrogen content in the irrigation water is nearly enough for the need of most crops and, therefore, no nitrogen fertilizers are needed. Furthermore, the correlation between the nitrogen load at the soil surface and the nitrate concentrations in the groundwater is investigated.

KEYWORDS

Agriculture, fertilizer, groundwater, manure, nitrate, sewage

INTRODUCTION

The Gaza Strip, see Figure 1, is a coastal area along the eastern Mediterranean Sea, about 40 km long and between 6-12 km wide. The area of the Gaza Strip is about 365 km^2 and it is densely populated with an estimated population of about one million inhabitants. The main source of water supply in the Gaza Strip is groundwater. The water quality of this resource is currently deteriorating in terms of increasing nitrate concentration and salinity content. Nitrate concentrations of more than twelve times the international accepted limits have been reported in some areas.

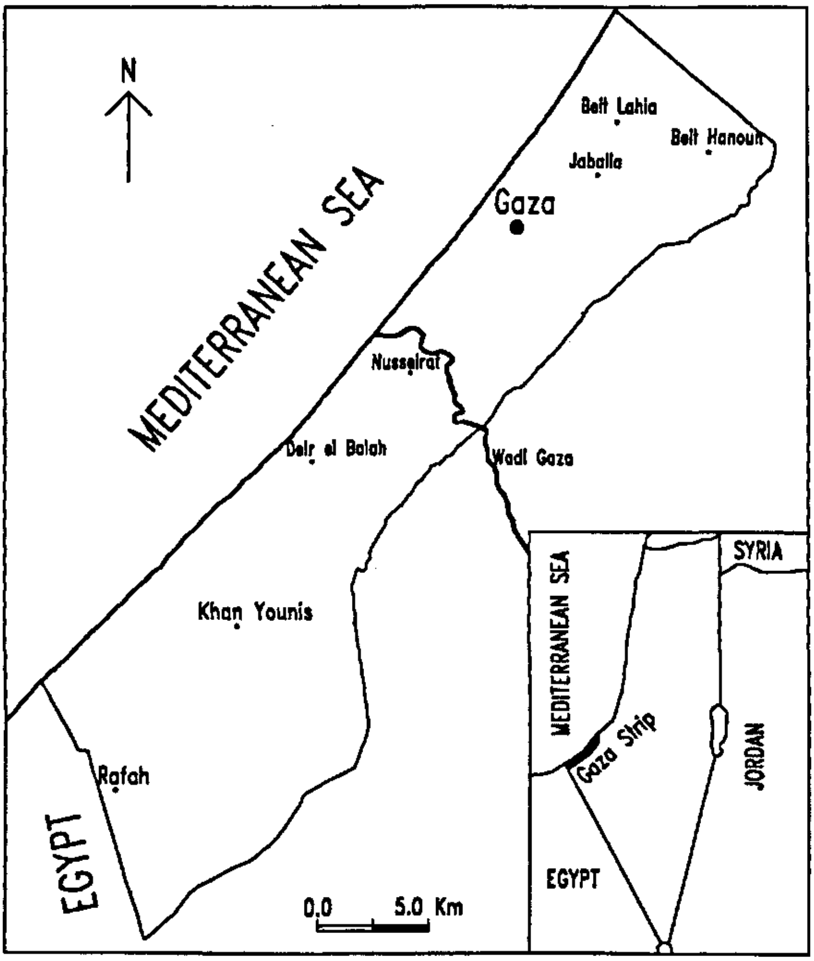


Figure 1. Map of the Gaza Strip

Water use in the Gaza Strip is mainly for domestic water supply and for irrigation. One third of the area of the Gaza Strip is used for agriculture. An estimate of the total water consumption for irrigation is 80 million m^3 per year. The entire amount is pumped out of the groundwater reserves by about 2000 registered agricultural wells, and by an estimated 1500 unregistered agricultural wells. Mineral fertilizers, manure and irrigation water are applied in excessive amounts. Nitrate rich leachate infiltrates to the groundwater. Most of the soil is sandy and the groundwater depth is shallow (about 60 m) which help the pollutants to reach the groundwater easily.

Domestic water supply (including network leakage and unaccounted for water) in the Gaza Strip is about 115 liters per capita per day. With a total population of about one million, the demand for domestic water in the Gaza Strip is estimated at 42 million m^3/year pumped by about seventy public water supply wells. Water used by the industrial sector is relatively small. Only one third of the Gaza Strip is covered with a sewage collection system. The majority of households have latrines connected to unsealed vaults. Therefore, a proportion of the liquid waste is conveyed through open brickwork and via seepage to the surrounding subsoil; the more solid septage is removed by vacuum tankers and discharged at the nearby soil surface (GEP, 1994, Mukhallalati *et al.*, 1995b).

NITRATE POLLUTION

Nitrate is one of the few pollutants of the groundwater in the Gaza Strip that are reasonably well documented. Typical nitrate concentrations are in the range of 100-200 mg NO_3 /l, with extreme values up to 650 mg NO_3/l (EPD, 1995, GEP, 1995, Mukhallalati and Safi, 1995*a* and PHG, 1994). The highest concentrations are observed in the densely populated areas with poor sanitary facilities. These concentrations largely exceed the WHO standards of 50 mg NO_3/l, and are a health threat especially for babies and pregnant women. The most important reasons for nitrate pollution appear to be the lack of proper sewage collection, treatment and disposal system and the excess use of fertilizers and pesticides in agriculture.

Two trends of nitrate concentration increase could be distinguished in the Gaza Strip. The first trend prevails in the rural agricultural areas with a gradual and low increase in nitrate, as represented by the first well in Figure 2. The nitrate concentrations are still low due to the relatively recent intensive agricultural activities. The second trend predominates in the urban areas and identified with a rather fast increase in nitrate, as illustrated by the second well in Figure 2. The nitrate concentrations in the urban areas are high in general as the area was inhabited long time ago.

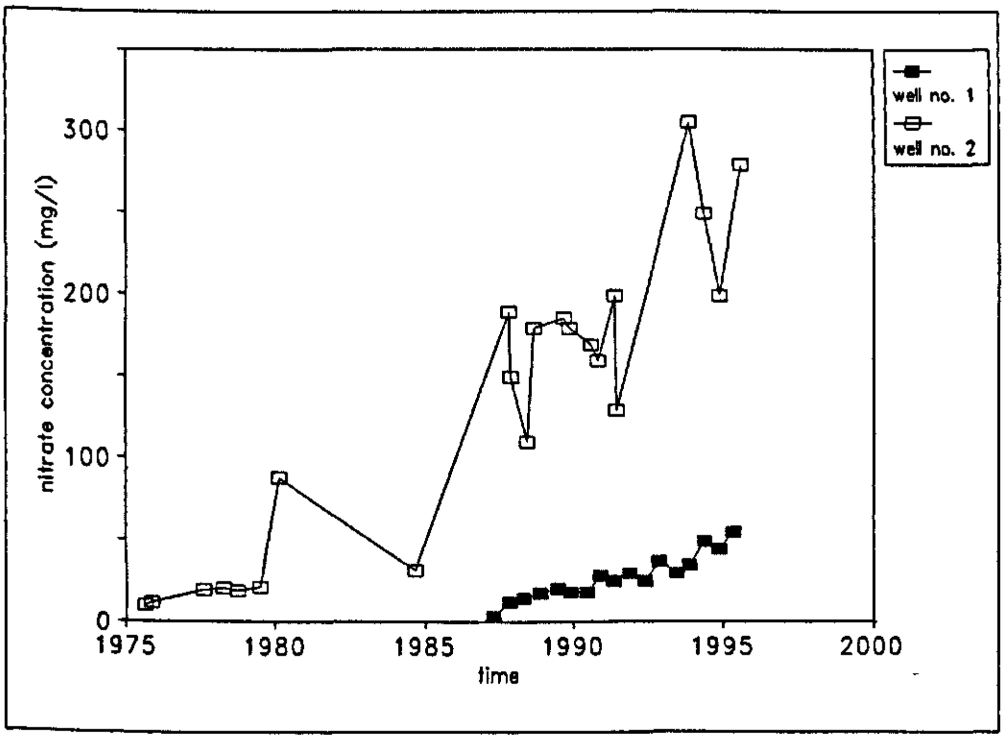


Figure 2. Nitrate concentration trends for two selected wells

CHEMICAL PROCESSES AND NITROGEN CYCLE

Nitrogen can be introduced to the soil/water system in different forms and at different places. The most important nitrogen sources in the Gaza Strip are manure and mineral fertilizers in the rural agricultural areas and human excreta in the urban areas. As a result of chemical and microbiological processes, nitrogen can be transformed into different forms. Nitrogen exists in the soil and the groundwater predominantly as the nitrate (NO_3^-) anion or as the ammonium (NH_4^+) cation. Which of the two prevails depends on the source of nitrogen, redox potential, soil pH, temperature, and so on. The nitrogen cycle is depicted in Figure 3.

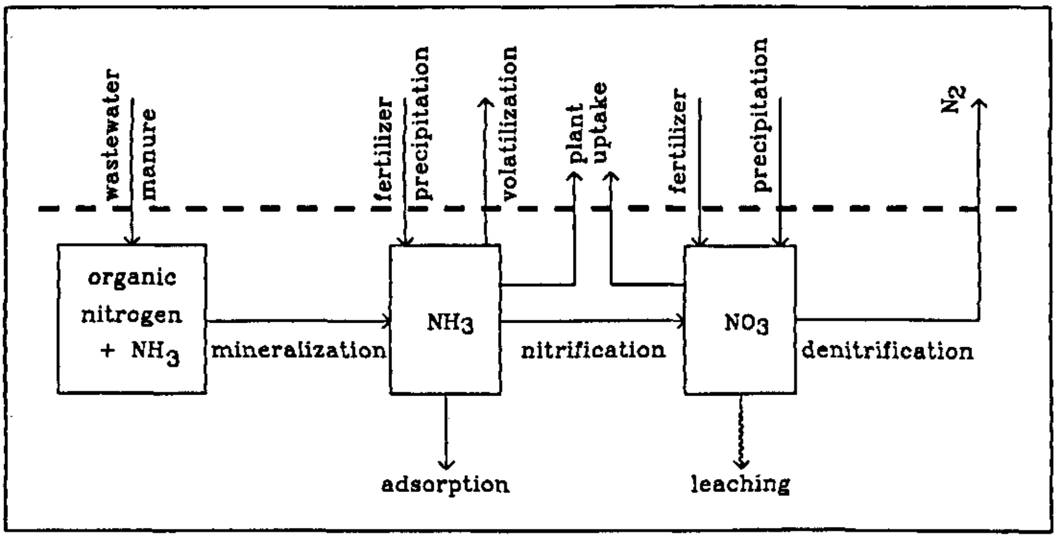


Figure 3. Schematic representation of the nitrogen cycle

Stable nitrogen compounds such as manure and human excreta can be transformed into less stable organic compounds and then into ammonium. This transformation process is microbiological and is called mineralization. A chemical (equilibrium) reaction between ammonium and the gas ammonia, may result in the volatilization of nitrogen. When oxygen is abundant, ammonia is transformed into nitrate either chemically or microbiologically by the nitrification process. The transformation is usually fast with a typical half life of one week. In soil systems, a part of the ammonium is adsorbed to the soil particles.

Nitrate is transformed into nitrogen gas in the denitrification reaction. It is a transformation process in which nitrogen is removed from the soil and/or the water system and is transferred to the atmosphere. This denitrification reaction is a microbiological process which only occurs under anaerobic conditions. During the reaction, organic matter is oxidized using nitrate as an electron acceptor. So, for the removal of nitrogen out of the soil/water system, nitrogen should first be nitrified (under aerobic conditions) and subsequently denitrified (under anaerobic conditions). This sequence of conditions is rare in the Gaza Strip where aerobic conditions in the unsaturated zone dominate.

METHODOLOGY

Nitrogen Balance Approach

The nitrogen sources and sinks, and the processes described in the previous section determine the nitrogen balance in the rural agricultural areas and the urban areas. The time period for nitrogen balance is taken as one year and the reference level for sinks and sources is taken as the ground surface. The nitrogen balance results in an N-excess or N-deficit. Giving insight in the flow of nitrogen, nitrogen balances are a powerful tool in identifying potential sources of nitrate pollution, and in helping to formulate appropriate remedial measures.

Nitrogen Balance for Agricultural Areas

The main components in the agricultural nitrogen balance in the Gaza Strip are; i) N-application in mineral fertilizers ii) N-application in organic manure iii) N-application by nitrate-rich irrigation water; and iv)N-uptake by plants. The other components that contribute to the nitrogen balance such as nitrogen losses through denitrification, volatilization, nitrogen storage and atmospheric deposition are marginal in the Gaza Strip and neglected for simplicity reasons. Field experiments in The Netherlands have shown that even after a long term application of manure, the nitrogen storage was rather limited. They have also shown that after four years without any application of manure no effect of an extra mineralization from the "stored nitrogen" on plants could be determined. Mineralization of manure is expected to be high in the semi-arid climate of the Gaza Strip because of the high soil temperature.

Soil conditions (eg. sandy or clayey), type of crop, irrigation method, quality of irrigation water, and climate may influence the N-balance considerably. Therefore, N-balance has been carried out separately for different locations, crops and soil types. The procedure is the same for each case, which could be summarized by calculating the nitrogen sources and sinks and then the nitrogen excess or deficit. The N-balance for strawberries which grow in sandy soil in Beit Lahia at the north of the Gaza Strip is given here as an example. One crop of strawberries per year is usually grown on the same plot for many years. The nitrogen components for the nitrogen balance are evaluated below.

(i) mineral fertilizers: Four N-containing types of fertilizers are used for the cultivation of strawberries as follows:

Table 1. Fertilizer application in strawberries

Type of fertilizer	N-content (%)	Application (kg/dunum)	Application (kg N/dunum)
Ammonium sulphate	21	26	5.5
Urea $CO(NH_2)_2$	46	30	13.8
Potassium nitrate	13	60	7.8
Compound 20-20-20	20	50	10
Total			37.1

(ii) Manure: Two types of manure are used for the cultivation of strawberries: cattle and poultry manure. The dry matter content of this raw manure is approximately 50%, while the N-content on dry matter basis is approximately 2.2% for a manure consisting of 50% cattle and 50% poultry manure (Bar-Yosef, 1995; personal communication). Based on the Agricultural Compendium (Euroconsult, 1989), this value is split up in 1% for cattle and 3% for poultry manure.

Table 2. Dry weight manure application in strawberries

Type of manure	N-content (%)	Application (kg/dunum)	Application (kg N/dunum)
Dairy manure	1	2000	20
Poultry manure	3	2000	60
Total			80

(iii) Irrigation water: The nitrogen concentration of the irrigation water is about 100 mg NO_3/l. With an estimated irrigation water of 1000 m^3/dunum/year, the nitrogen application could be calculated as 23.4 kg N /dunum/year.

(iv) Plant uptake: The plant uptake of strawberries is approximately 6-10 kg N/dunum/year. The nitrogen uptake could be taken as 10 kg N/dunum/year because of the high nitrogen and irrigation water application (Bar-Yosef, 1995; personal communication). The nitrogen balance for strawberries can be calculated as shown in Table 3.

Table 3. Nitrogen balance for strawberries (Beit Lahia)

Component	N-source (kg N/dunum)	N-sink (kg N/dunum)
Fertilizer	37.1	
Manure	80.0	
Irrigation water	23.4	
Plant uptake		10.0
Total	140.5	10.0

Having a look at the results presented in Table 3., it is clear that the N-sources exceeds the N-sinks by 130.5 kg N/dunum which represents the nitrogen excess. An interesting result is that the nitrogen content in the irrigation water is more than double the nitrogen uptake by strawberries, which means that irrigation water contains enough nitrogen for the need of strawberries and that there is no need to add any nitrogen fertilizers.

Nitrogen Balance for Urban Areas

Domestic wastewater is considered the major non-point source of nitrogen in urban areas in the Gaza Strip. In fresh wastewater, nitrogen is present in the form of organic nitrogen combined in proteinaceous matter and urea. Only 30% of the Gaza Strip is provided with a sewage collection system. The rest of the Gaza Strip is unsewered and depends on percolating pits or cesspits to collect and "treat" the sewage. Nitrate rich leachate from the cesspits infiltrates into the unsaturated zone and may percolate to the groundwater.

The nitrogen balance in the urban areas of the Gaza Strip is mainly affected by the population density and the percentage of sewage which is evacuated from each community (city, town, village, etc.). Therefore, the nitrogen balance is carried out for each community separately. The nitrogen balance approach for the urban areas assumes that the population is uniformly distributed within the area of each community. In this article, the calculations of the nitrogen balance for Gaza City is given as an example.

The major nitrogen source, which is originated from sewage, could be calculated for each community by knowing the number of population and the built up area. N-production by human excreta is estimated at 10 g N/cap/day (Pescod, 1992). For example, the number of population in Gaza City is 289 300 and the built up area is 23520 dunum. This results in a population density of 12.3 cap/dunum, and a nitrogen production of 123 g N/dunum per day or 44.9 kg N /dunum per year.

Plant uptake of nitrogen is negligible in the urban areas. First of all, there is no much plant growth in the urban areas. Secondly, plants are not able to take up nutrients from a depth of 6 to 10 m below ground level (the bottom of most percolating pits). N-storage may occur in the form of sludge but it contains the more stable nitrogen compounds and, therefore, it is neglected for simplicity. The major nitrogen sinks are:

(i) Physical nitrogen removal through evacuation of sewage by the sewage collection network. For example, 40% of the sewage in Gaza City is collected and transmitted by the sewage network. The removed amount of nitrogen is thus: 40% x 44.9 kg N/dunum = 18.0 kg N/dunum.

(ii) N-removal through denitrification and volatilization in cesspits and open drains. This is estimated at 20% of the nitrogen which is not transmitted. For example, nitrogen amount which is not transmitted in Gaza City is 44.9 - 18.0 = 26.9 kg N/dunum. Thus, N-removal is calculated like this: 20% x 26.9 kg N /dunum = 5.4 kg N/dunum. The nitrogen balance for Gaza City is given in Table 4.

Table 4. Average nitrogen balance for Gaza City

Component	N-source (kg N/dunum)	N-sink (kg N/dunum)
Human N-production	44.9	
Evacuation of sewage		18.0
N-removal in cesspits		5.4
Total	44.9	23.4

The results of the nitrogen balance show that N-source exceeds N-sink by 21.5 kg N/dunum. The results of the other nitrogen balances carried out for all the communities in the Gaza Strip indicate nitrogen excess. The amount of nitrogen excess varies from one community to another according to the change in population density and the percentage of sewage evacuated by the sewage collection system. The nitrogen excess leaches to the groundwater causing high nitrate concentration .

DISCUSSION AND CONCLUDING REMARKS

The nitrogen components in the rural agricultural areas and the urban areas in the Gaza Strip have been assessed. A more accurate assessment may be achieved by evaluating the neglected components and processes which contribute to the nitrogen balance. The results of the nitrogen balances show that there is a nitrogen excess mainly because of the use of excessive amounts of fertilizers and irrigation water in the agricultural rural areas, and because of the lack of sewage collection, transmission and treatment system in the urban areas.

The nitrogen excess in the area cultivated with strawberries is estimated at 130.5 kg N/dunum/year. The recharge rate at the same area is calculated as 670 mm/year. Assuming that all the nitrogen excess is transformed into nitrate, a simple calculation results in a nitrate concentration of 833 mg NO_3/l in the leachate. The nitrate concentrations in the leachate calculated for citrus grown in clayey soil in Beit Hanoun and for potatoes grown in sandy soil in Beit lahia are 258 and 382 mg NO_3 /l, respectively. These values are a factor of 2 to 4 higher than the measured concentrations in the water samples extracted from the agricultural wells. The reasons beyond such a deviation are mainly due to the following:

(i) There is a dilution factor in groundwater abstracted from agricultural wells. An agricultural well abstracts groundwater that derives from a larger area than just the one dunum of strawberries. If the adjacent lands are fallow the recharging rainwater there will be low in nitrates.

(ii) There is a time delay between nitrogen application at the soil surface and a concentration increase in the groundwater. So, given the fact that the intensive horticulture developed mainly during the last 10 years, the calculated nitrate increase could "still be on its way" (Ronen *et al.*, 1983).

The results of the nitrogen balance calculations for Gaza City results in nitrogen excess of 21.5 kg N/dunum/year. The recharge from rainfall plus the percolation of wastewater that is not removed by sewage system is estimated at 449 mm/year. If all the nitrogen transforms into nitrate then the nitrate concentration in the leachate will be 205 mg NO_3/l. The calculated values for the other urban towns and villages range between 200 to 300 mg NO_3/l, and about 500 mg NO_3/l in the refugee camps. In general, there is a good agreement between the calculated concentrations and the measured concentrations in the water samples collected from the urban areas. The reason is that the previously mentioned dilution factor and time delay in the urban areas are much less effective than the rural agricultural areas. The urban areas are relatively large and the nitrogen load is uniformly distributed over the whole area of a considered community. Also, the urban areas are inhabited many years ago which means that the unsaturated zone is rather saturated with nitrogen. Moreover, the nitrogen is released at a typical depth of 6 to 10 m below ground level (the bottom of most percolating pits). This means that a considerable part of the unsaturated zone is by-passed, particularly the top soil where microbial activity is concentrated.

ACKNOWLEDGEMENT

This study has been supported by the Government of The Netherlands and carried out by a team of experts from the Environmental Planning Directorate, Ministry of Planning and International Cooperation, the Palestinian National Authority together with experts from IWACO and EUROCONSULT, The Netherlands.

REFERENCES

Euroconsult, 1989. Agricultural Compendium for Rural Development in the Tropics and Subtropics.

EPD (Environmental Planning Directorate) and Euroconsult/Iwaco (1995) "Nitrate Pollution Study in the Gaza Strip".

GEP (Gaza Environmental Profile) (1994), Part One, "Inventory of Resources", Palestinian Environmental Protection Authority, Euroconsult/Iwaco.

GEP (Gaza Environmental Profile) (1995), Part Two and Three, "Relation between Man and Environment, Problems and Solutions", Palestinian National Authority, Euroconsult/Iwaco.

Mukhallalati, L., and Safi, J., (1995*a*) "Nitrate Pollution of Groundwater in the Gaza Strip", Proceeding of the Symposium on A Strategy for Water Sector Capacity Building in Palestine, pp. 137-140, 6-7 September 1995, Birzeit, The West Bank.

Mukhallalati,L., Ajjour,M., Gischler,M. and Koopmans,R., (1995*b*)"Water scarcity and quality deterioration in Gaza", Proceeding of the Second International Conference on the Mediterranean Coastal Environment, MEDCOAST 95, pp. 1373-1380, 24-27 October 1995, Tarragona, Spain.

PHG (Palestine Hydrology Group) (1994), "Water Quality Data on Public Water Supply Wells, Gaza".

Pescod, M. B., 1992. Wastewater Treatment and Use in Agriculture. Irrigation and Drainage Paper No. 47, FAO, Rome.

Ronen, D., Y.Kanfi, M. Magaritz, 1983.Nitrogen Presence in Groundwater as Affected by the Unsaturated Zone. In Pollutants in Porous Media; the unsaturated zone between soil surface and groundwater. Ed. B. Yaron, G. Dagan, J. Goldschmid.

Volcani Institute, Institute for Soil and Water. Data on N-uptake by plants (personal communication with Dr. Bar Yosef).

NITROGEN DYNAMICS IN BUFFERSTRIPS ALONG LOWLAND STREAMS.

Jeroen J.M. de Klein & Mariet M. Hefting

Department of Water Quality Management and Aquatic Ecology
Wageningen Agricultural University
P.O. Box 8080
6700 DD Wageningen
The Netherlands

ABSTRACT

Surface waters in the Netherlands are polluted with nutrients from agricultural land. Transitions from agricultural land to surface waters are usually sharp. Bufferszones can make the transition more gradual. Processes as denitrification and accumulation in biomass which occur in bufferzones can reduce the load of nitrogen to streams and brooks.

We studied nitrogen transport and transformations in a riparian forest strip and a grassland buffer zone. In the period October 1996 to June 1997 measurements were done on groundwater level and groundwater quality, surface water discharge and quality and denitrification activity in bufferzones

We measured nitrogen transport and processes in a headwater of a small natural brook, the Hazelbeek in Twente. On the northern side of this stream a extensively used grassland and arable land (winter wheat) is situated. On the south side the research area is adjacent to a field of maize with high production and fertilisation levels.

The transition from agriculture land to the brook is formed by a hilly zone of 25 to 55 meters wide. The bufferzone consists of a combination of riparian forest (alder thicket) and grassland. Two rows of groundwater tubes were placed at right angels to the stream and sampled 11 times. In the transition zone gaseous nitrogen fluxes (N_2O) were measured using an photoacoustic tracegasanalyser. Denitrification in soilcolumns was measured at the laboratory using the acetylene inhibition method.

During all measurement series we found clear spatial differences in groundwater quality. High nitrate concentrations (45-50 mg N/l) were found in groundwater tubes placed on the boundary of the agricultural land (maize) and the grassland strip, lower concentrations were found at the boundary of the extensive agricultural field. Quality measurements in surface water indicated that surface flow from the agricultural land strongly contributed to the nitrogen load in the brook. Denitrification rates measured in the top soil of the bufferzones varied both spatial and temporal between 5 and 600 kg N/ha/yr.

Measurements are continued to determine the efficiency of bufferzones in the reduction of nitrogen loads to streams and brooks.

KEYWORDS

agricultural runoff; bufferzones; denitrification; groundwater; riparian forest; surface water

INTRODUCTION

Diffuse pollution of surface waters with nutrients from agricultural soils can be seen as a major environmental problem for watermanagement in the Netherlands. About 60-90% of the total discharge of

nitrogen to surface waters is originating from agricultural areas. High concentrations of nitrate in surface waters leads to eutrophication and in turn to instability of the aquatic ecosystem .
Reduction of the amount of fertiliser used on agricultural fields will probably have its effects on the longer time span. Therefore measures which directly influence the nutrient emission to surface waters need to be studied.
Past research'had shown that bufferzones as riparian zones or grassland strips are effective in removing or assimilating nutrients entering from upslope agricultural fields via shallow lateral flow. The reduction in nutrient load to streams and brooks can be attributed to a combination of processes including denitrification, nitrogen accumulation in biomass and reduction of sediment content in runoff.

OBJECTIVE OF THE STUDY

The objective of this study was to quantify the nitrate removal in two vegetation types of riparian zones and to evaluate the contribution of denitrification activity in the nitrate removal. Relationships between organic matter, soil water content and denitrification activity were also investigated.

NITROGEN TRANSPORT AND TRANSFORMATIONS IN BUFFERSTRIPS

Definition of bufferstrips

A bufferstrip (also called vegetated filter strip VFS) can be defined as a transition zone between agricultural fields and surface waters which can remove sediment and nutrients from surface runoff and subsurface flow. These strips potentially can serve to reduce the diffuse pollution of surface water with nutrients. The nutrient removal from surface runoff and subsurface flow by buffer zones can be attributed to two mechanisms:

• Physical removal of particle bound nutrients

Vegetation on bufferstrips reduces the velocity of the runoff flow resulting in an increased infiltration and consequently deposition of coarse particles. Furthermore suspended particles are filtered through leaf litter and the soil. An important factor for bufferzones is the equal distribution of surface runoff over the bufferstrip. When channelling occurs removal of nitrogen from surface runoff is less efficient (Norris, 1993)

• Biological removal of nutrients

Bufferstrips can facilitate a variety of biological transformations to reduce the nutrient load of surface and subsurface flow: plant uptake; microbial immobilisation; nitrification and denitrification (Lowrance et al 1984).
A number of studies have identified biological denitrification as a process which might contribute significantly to nitrate reduction in riparian soils (Peterjohn and Correll 1984, Ambus and Lowrance 1991, Ambus and Christensen 1993). The dynamics and control of denitrification in these ecosystems in the Netherlands has only been investigated to a limited extent (Orleans, 1994).
Because biological processes are important in the removal of nutrients from surface runoff and subsurface flow, the efficiency of these zones is not only dependent on the type and size of the strip but varies with physical, chemical and environmental conditions. Comparison of results from literature with one particular site in the Netherlands is not always strait forward.

Leaching of nitrate

In the Netherlands the rate of fertiliser application on grassland and arable land is high.

In this research focus lies on the reduction of the nitrogen load to surface waters. Nitrate can easily be lost from the soil system by leaching, because of its high solubility and the negative charge of the ion, which limits its adsorption to soil particles. Losses of nitrate by leaching increase in the autumn and winter period, when uptake of nitrate by the crop is negligible and water movement is mainly downward because of the precipitation surplus. Important factors that control nitrate leaching are: soil type, ground water level, crop growth and the amount of fertiliser applied.
According to the EU drinking water standards nitrate concentration in ground and surface water should not exceed 50 mg/l. This standard concentration is often exceeded.
With regard to the retention or reduction of nitrate denitrification is an important mechanism.

Denitrification
Denitrification is a potentially important process for nitrate removal in bufferstrips. Denitrification is desirable process in bufferstrips because of the complete removal of nitrate from the bufferstrip, whereas nitrate that is taken up by plants or is immobilised in microbial biomass can be mineralised and released to the soil solution (Groffman et al 1991).
The dominant end product of denitrification can be nitrous oxide or nitrogen gas depending strongly on the soil conditions. Under microbial suboptimal conditions the dominant product will be nitrous oxide. Nitrous oxide is also dominant in case of high nitrate concentrations in the soil solution. Under anaerobic conditions with less nitrate available, nitrogen gas will be the dominant end product. The amount of nitrous oxide produced from denitrification and nitrification is important because nitrous oxide emissions are increasingly recognised as an important factor contributing to the anthropogenic greenhouse effect and climate change (Bouwman, 1990).

Factors controlling denitrification
Three factors are essential for denitrification; available nitrate, available carbon and anoxic conditions.
• NO3
Nitrate availability: In natural forests and grasslands the nitrate availability depends on the mineralization rate combined with the nitrification activity. In bufferzones bordering intensively managed agricultural fields, nitrate availability will often be high enough to support denitrification activity.
• O2
The oxygen concentration is dependent on the soil diffusivity and oxygen consumption which are influenced by several soil factors such as soil water content, texture, organic matter and pore size distribution. Riparian soils are often waterlogged or have temporal high groundwater levels.
• C
Often wet or saturated bufferzones contain high levels of organic matter. Generally these conditions favour denitrification. It depends however on the quality of the organic matter. The amount of available carbon or C/N ratio of the organic matter appears to be important (Groffman et al 1991).

MATERIALS AND METHODS

Site description
We studied nitrogen transport and transformations in a riparian forest strip and a grassland buffer zone. In the period October 1996 to June 1997 measurements were done on:
- groundwater level and groundwater quality
- surface water discharge and quality
- denitrification activity and nitrous oxide fluxes from the bufferzones

Two sites were monitored, first a grass zone with permanent unfertilised pasture of 0.3 ha (max. 25 m wide) adjacent to an arable field grown with winter wheat. Secondly an alder thicket zone of 1.6 ha (max. 55 m wide) bordering an extensively managed grassland and intensively managed arable land with maize.

The alder thicket zone is situated downvalley of the grassland zone (see fig. 1.). The sedimentary structure was largely comparable between the two zones. The slopes of the riparian zones are comparable.
A transect of groundwater tubes was placed in both zones at right angels to the stream (see fig. 2). The depth off the groundwater tubes ranged from 5.5 to 1.8 m. Because groundwater tubes are not consequently placed above the impermeable layer of tertiary clay interaction with deeper groundwater can not be ruled out.

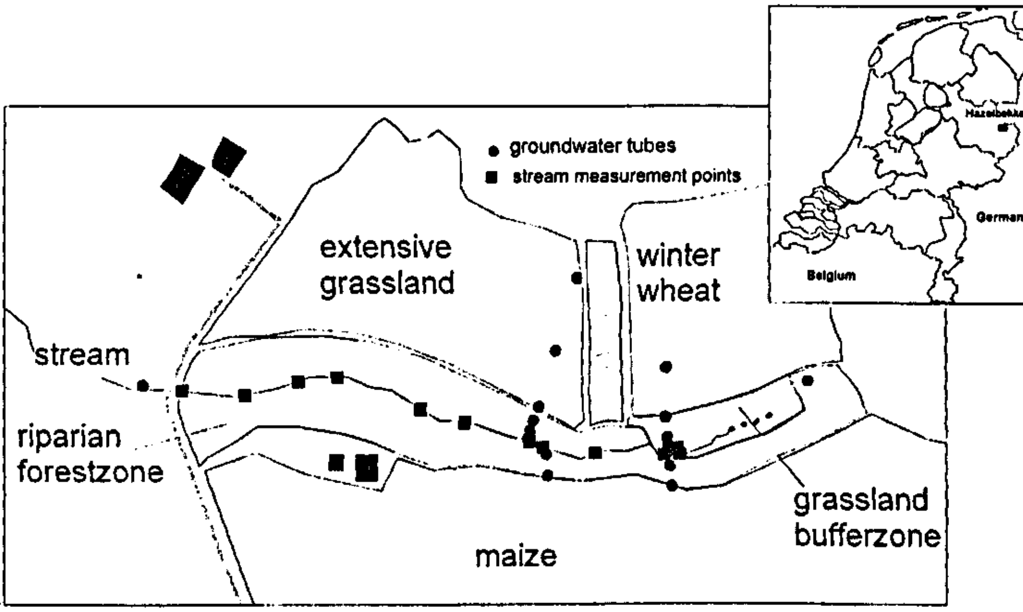


Fig.1. Overview research area Hazelbekke.

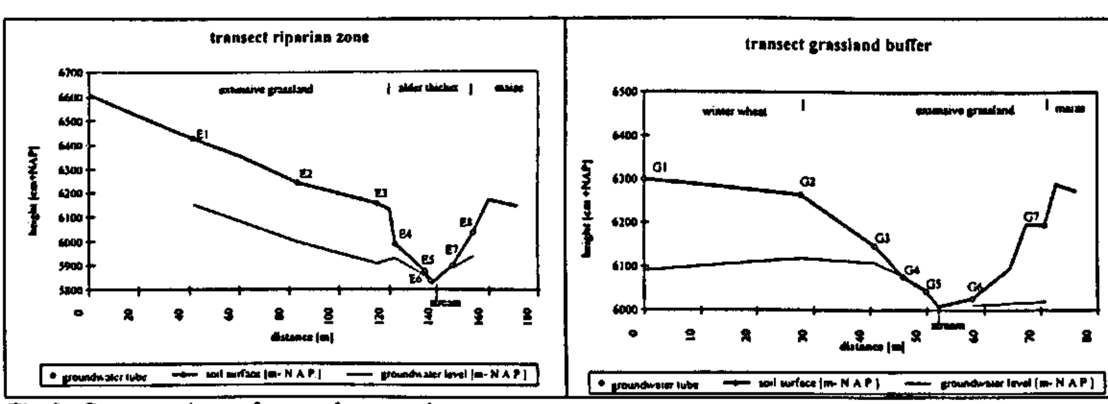


Fig.2. Cross sections of groundwater tubes transect.

Sampling

Water samples from the brook were taken by hand in October, November, January, April and June at 12 locations distributed along a 300 m section of the stream. Water samples from the groundwater tubes were taken with a *pulse pump*. Samples were stored at 5°C and analysed for nitrate and nitrite -N and ammonium-N on a Skalar autoanalyser. Furthermore chloride concentrations and EC were measured in the samples.

Although denitrification activity and vegetation uptake are thought to be insignificant with temperatures below 5°C, the bulk of the measurements is executed in the autumn and winter month because in temperate regions leaching of nitrate from agricultural soil is dominantly occurring in these month.

Denitrification experiments

In the transition zone gaseous nitrogen fluxes (N_2O) were measured using an photoacoustic tracegas-analyser (Bruel and Kjear 1302). Denitrification in soil columns was measured at the laboratory using the acetylene inhibition method. Denitrification was measured using the soil core technique and acetylene inhibition technique described by Tiedje (1982). Cores with an average length of 30 cm and 5 cm diameter were removed from soil in Plexiglas tubes. The tubes were sealed with rubber stoppers. In the laboratory nitrous oxide fluxes were measured before and after addition of 10 kPa acetylene. To allow dispersion of acetylene in the soil pores measurements took place 2 hour after acetylene injection.

RESULTS

Groundwatertubes

Nitrate concentrations in the groundwatertubes do not vary much in time and show a quiet stable spatial pattern. In figure 3 and table 1 this is presented. The highest concentrations of NO_3 are found at the edge of the high production maize field (>40 mgN/l). The lowest concentrations are found in the tubes close to the brook, in the centre of the riparian forest strip.

Table 1. Nitrate concentrations in groundwater

MPCODE	AVG * (mgN/l)	STD * (mgN/l)
e1	0.39	0.43
e2	0.15	0.08
e3	6.26	1.93
e4	1.66	1.18
e5	0.26	0.26
e6	0.15	0.15
e7	0.07	0.04
e8	0.54	0.63
g1	3.26	0.58
g2	12.15	1.47
g3	6.02	2.46
g4	8.02	2.29
g5	1.92	2.04
g6	1.42	1.06
g7	40.27	4.95

* 11 samples

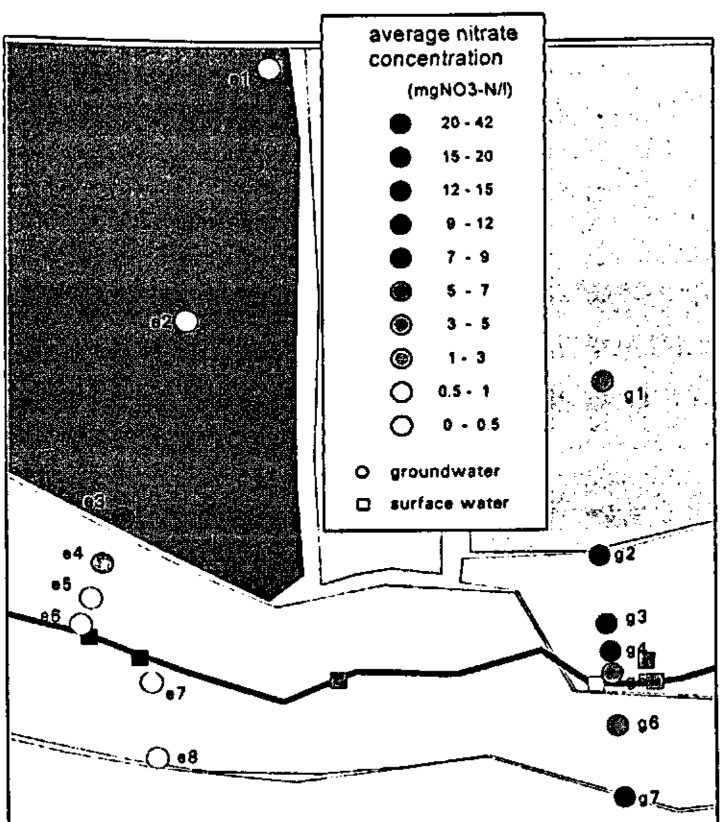


Figure 3. Spatial pattern of average NO3-concentrations.

The results show a clear drop in nitrate concentration in the groundwater from the edge of the bufferzone to the stream. In the transect e3-e6 average concentrations drop from 6.2 to 0.15 mgN/l and in the transect g7-g5 from 40.3 to 1.9 mgN/l.
The results from the tubes outside the bufferzone (e1,e2,g1) are lower then at the edges, which cannot be explained. As stated before it might not be ruled out that these tubes are in contact with other (deeper) groundwater layers. Analysis of chloride-concentrations of the samples show that this might be the case.

Surface water quality

Like the groundwater results the concentrations of nitrate in the stream surface water show a more or less stable spatial pattern (fig. 4). There is a remarkable peak concentration at sampling point H5. The concentrations are high compared to the phreatic groundwater close to the stream (e6,e7, g5). We observed a clear erosion channel in the slope of the bufferzone from the maize field to the stream located at H5. Although surface runoff at this spot is not observed it is likely that surface runoff and rapid subsurface runoff may cause the peak concentration in the stream. The local geohydrologic situation is not yet investigated. Downstream (to H12) nitrate concentrations drop by instream of 'cleaner' groundwater.

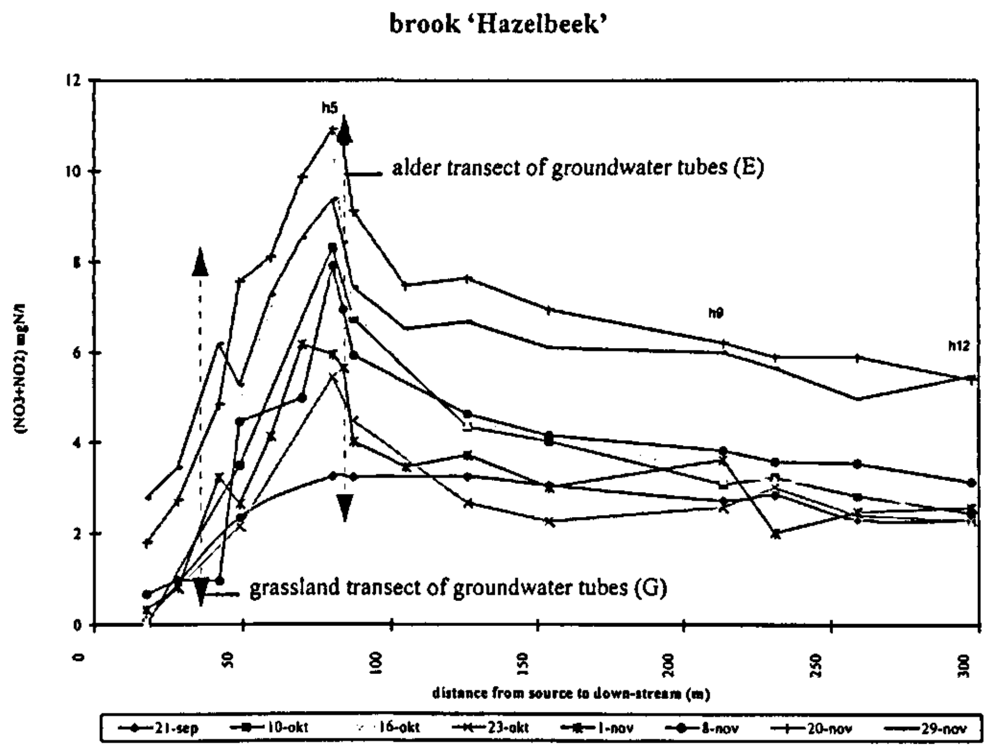


Figure 4. Longitudinal profile of nitrate in the stream surface water

Denitrification rates

An average rate of 116 kgN/ha/y was measured in the riparian forest strip. The rates of denitrification vary both spatially and temporal (see table 2.).
Lowrance et al. (1984), Peterjohn and Correl (1984) all found denitrification rates around 30 kgN/ha/y in riparian forests. Also rates from 100 to 200 kgN/ha/y are reported (Ambus and Lowrance, 1991; Kruijne, 1996).

Table 2. Denitrification rates in the bufferzones.

Code	Vegetation type		**Denitrification rate [kg N/ha/yr]**		
		01-11-96	08-11-96	29-11-96	10-06-97
E I	alder	28.6	0.2	4.3	0,2
E II	alder	724,6	50,4	20,9	-0,8
E III	alder	50	99,4	235	33,4
G I	grass	61,4	4,8	14,6	0,7
G II	grass	2,8	14,2	4,3	1,6
G III	alder	53,7	6,5	99,5	3

DISCUSSION

We found that denitrification fluxes in the forested zones are generally high compared to the fluxes from the grassed zone. This can indicate a higher organic matter quality in the forested zones.
This result is in contrast to the denitrification fluxes found by Groffman et al (1991) and Schnabel et al (1996). They observed higher denitrification fluxes in the grassland bufferzones compared with the forested bufferzones. An increase in denitrification activity was observed when soil samples from forest buffer strips where amended with glucose thus indicating that organic matter quality was less favourable in the forested strips.
Denitrification is a highly variable process. Coefficients of variation are between 100-300%. In this research no replicates were taken for denitrification measurements.
From results on denitrification and nitrous oxide fluxes (not shown) in the field and the laboratory it can be concluded that the contribution of nitrification to the nitrous oxide flux is not to be neglected because nitrous oxide fluxes under unamended conditions can be higher than the nitrous oxide flux under acetylene amended conditions.

A problem that has emerged from riparian zone research is that when denitrification and immobilisation have been measured directly in a laboratory set-up, the measured rates are frequently too low to account for the amount of nitrate removal from groundwater in riparian forests (Groffman et al , 1996).
This can be caused by the high spatial and temporal variability of denitrification (denitrification in hot spots), differences in scale between field measurements and laboratory measurements, and denitrification by chemoautotrophic bacteria using pyrite. In our study denitrification in the subsoil was not quantified.
Although vegetation has no active role in retaining nitrate in winter , above ground vegetative biomass does contribute C to the soil microbacterial biomass that is engaged in nitrate reduction in winter month. This accounted for the greater efficiency of the poplar vegetated site compared to the grass vegetated riparian site (Haycock and Pinay, 1993).

Because Cl is not actively taken up by organisms it often can be used as a conservative marker. Osborne and Kovacic 1993 found significantly lower nitrate-N:Cl ratio's at all depth in bufferstrips. These ratio's support that denitrification is an important mechanism for the removal of nitrate-N from groundwater.
In our study chloride concentrations were monitored in all water samples. Concentrations showed large variations probably originating from the tertiary clay in the subsoil which was formed under marine conditions.

CONCLUSIONS

We found that nitrate concentrations in groundwater decrease with 95% in bufferzones (riparian forest).

In the bufferzone high rates of denitrification were measure that contribute to the reduction of nitrate concentration in groundwater. Channelled surface runoff from the maize field is a possible 'point-source' for the stream.

When rates and concentrations are extrapolated to a whole year and the whole research area (with some uncertainty, due to spatial and temporal variability) it is clear that N-export from the catchment by surface water discharge and by denitrification in the riparian forest are in the same range of magnitude (210 and 170 kgN/year respectively).

ACKNOWLEDGEMENTS

A major part of the field research was done by E. Schellekens for his MSc-thesis research.

REFERENCES

Ambus P. and Christensen S. (1993). Denitrification variability and control in a riparian fen irrigated with agricultural drainage water. *Soil biology and Biochemistry 25: 915-923*

Ambus P. and Lowrance R. (1991). Comparison of denitrification in two riparian soils. *Soil Sci Soc Am J. 55:994-997*

Bouwman A.F.,(1990). Soils and the greenhouse effect. *Proceedings of the international conference on soils and the greenhouse effect.*

Groffman P.M., et al., (1991).Denitrification in grass and forested vegetated filter strips. *J. Environ. Qual. 20:671-674.*

Groffman P.M., et al. (1996). Microbial nitrate processing in shallow groundwater in a riparian forest. *J. Environ. Qual. 25:1309-1316.*

Haycock, N.E. and Pinay G. (1993). Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *Journal of Environmental Quality 22: 273-278.*

Kruijne, R. (1996). Perceelsonderzoek naar het effect van beekbegeleidende bufferstroken op de stikstof- en fosforbelasting van de Mosbeek. Nutriëntonderzoek en scenarioberekeningen. *DLO-Staring Centrum, Wageningen. Rapport 420.2.*

Lowrance R.R, Todd R.L. and Asmussen L.E.(1984). Nutrient cycling in an agricultural watershed-I phreatic movement. J. Environ. qual.13:22-27.

Norris, V.O.L., (1993). The use of buffer zones to protect water quality: a review. *Water Resources Management 7: 257-272.*

Orleans, A.B.M.,. et al.., (1994). Minder nutriënten in het oppervlaktewater door bufferstroken? Een literatuuranalyse. *Milieubiologie R.U. Leiden.*

Osborne, LL & D.A. Kovacic, (1993). Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology 29: 243-258.*

Peterjohn, W.T. & D.L. Correl, (1984). Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology 65(5): 1466-1475.*

Schnabel, R.R., Cornish, L.F., Stout, W.L., and Shaffer, J.A., (1996). Denitrification in grassed and a wooded, valley and ridge, riparian ecotone. *J.Environ. Qual. 25:1230-1235.*

Tiedje J.M.(1982). Denitrification. *In: Methods of soil analyses. Part 2. Chemical and microbiological properties. 1011-1026. ASA-SSA. Madison USA.*

INFLUENCES OF THE HYDROLOGIC REGIME OF STREAMS ON THE EUTROPHICATION PROCESS IN LAKE NULDERNAUW AND LAKE WOLDERWIJD, THE NETHERLANDS

R.E. Rijsdijk, T.H. Helmerhorst and H.H. Reeders

Dutch Water Authority Directorate IJsselmeer Area, P.O. Box 600, 8200 AP Lelystad, the Netherlands

ABSTRACT

Lake Nuldernauw and Lake Wolderwijd (The Netherlands) are both shallow lakes with a small hydraulic residence time (30 - 90 days). The lakes receive water by streams, precipitation, seepage and inlet from Lake Veluwemeer. It appears that the phosphorus loads added by streams in the late winter (February - March) are regulating the phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd in spring (April - June). These streams are mainly loaded with phosphorus by agricultural runoff, since the greater part of the catchment area is phosphorus saturated. The contribution of the streams to the external phosphorus loading varies between 32% and 53%, in dependence to fluctuation in the precipitation surplus. As compared to dry periods, the phosphorus loads in wet periods added by streams are higher: the water flows are higher while the phosphorus concentration increases as well, due to a higher agricultural runoff. As a result, the phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd shows an erratic pattern in time.

KEYWORDS

agricultural runoff; eutrophication; hydrology; phosphorus; shallow lake; streams

INTRODUCTION

Lake Nuldernauw and Lake Wolderwijd, in The Netherlands (figure 1), were formed as marginal lakes after embankment of part of Lake IJsselmeer in 1967. Both lakes are shallow and are traversed by a shipping canal. The main physical characteristics of both lakes are shown in table 1.

Table 1. Main characteristics of Lake Nuldernauw and Lake Wolderwijd

	Lake Nuldernauw	Lake Wolderwijd
Mean water depth (m)	1.84	1.81
Water surface area (10^4 m^2)	701	1,854
Hydraulic residence time (days)	< 30	90

Lake Nuldernauw and Lake Wolderwijd receive water from streams, precipitation, seepage and inlet from Lake Veluwemeer. Discharge of water occurs by evaporation, groundwater infiltration, outlet to land and Lake Nijkerkernauw. The external loads are contributed by 'fixed' and 'variable' input terms. The 'fixed' input terms are more or less independent of season (seepage and inlet from Lake Veluwemeer), while the 'variable' input terms are dependent of season (streams and precipitation).

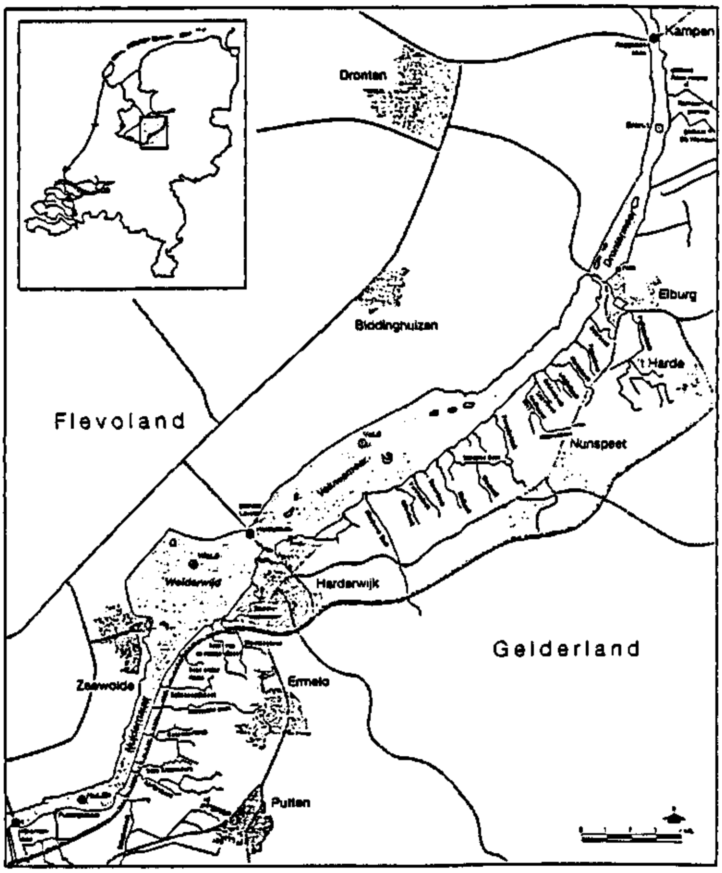


Fig. 1 Lake Nuldernauw and Lake Wolderwijd, The Netherlands

The contribution by the external phosphorus loads in the streams is substantial (table 2). The mean yearly contribution by streams to Lake Wolderwijd is one third of the phosphorus budget. The mean yearly contribution by the streams to Lake Nuldernauw is about half of the phosphorus budget. The relative contribution of the phosphorus loads by streams greatly vary yearly: 4% (1992) to 65% (1979) for Lake Wolderwijd and 21% (1992) to 77% (1987) for Lake Nuldernauw (Van Ballegooijen and Van der Molen, 1994).

Table 2. Yearly contribution (average 1976 - 1992) by streams to Lake Wolderwijd and Lake Nuldernauw (Van Ballegooijen and Van der Molen, 1994).

	Lake Nuldernauw	Lake Wolderwijd
water budget (%)	31	7
P budget (%)	53	32

Four streams discharge into Lake Wolderwijd (Stadsweidebeek, Weisteegbeek, Beek van de Hooge Geest and Beek onder Horlo) and six streams discharge into Lake Nuldernauw (Schaapsdijkbeekje, Horstschebeekje, Dasselaarbeek, Beek Krakenburg, Aartjesbeek and Schuitenbeek). The stream Schuitenbeek accounts for half of the loads (water and P) of all streams combined and therefore is considered to be the most important one in this area.

According to De Boer *et al.* (1996), water flows and contents of streams mainly depend on (geo)hydrology, physical, chemical and (vegetation)physiological processes and human activities (*eg* land use in the catchment area). The loads of phosphorus to streams happen in this area because of:

- Natural or background runoff: the natural concentration of streams in this area is about 0.06 - 0.08 mg P l^{-1} (PER, 1982);
- Sewer: about 0.9 10^3 kg P yearly (Reeders and Helmerhorst, 1996);
- Agricultural runoff: 80% of the catchment area of stream Schuitenbeek is phosphorus saturated (De Boer *et al.*, 1996).

The catchment area of the stream Schuitenbeek is about 8265 ha (De Boer *et al.*, 1996), mostly consisting of sandy ground, with 64% of the area having a high groundwater table (Breeuwsma *et al.*, 1989). It can be assumed that the loads from sewer and natural runoff are more or less constant over time. Loading from agricultural runoff is variable over time.

As algal growth in Lake Wolderwijd and Lake Nuldernauw is limited by the phosphorus concentration (Reeders *et al.*, 1997), this is the focus of this paper. We investigate to what extend the hydrologic regime of phosphorus rich streams determines the eutrophication process in Lake Nuldernauw and Lake Wolderwijd. The hypothesis that there is a (simple or multi-)relation between water flows, phosphorus concentration or phosphorus loads added by the streams in winter and the phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd in spring and summer is tested.

METHODS

From 1985 onwards, the water flows of stream Schuitenbeek were measured continuously by the Water Authority North Veluwe. The nutrient concentration was measured once a week by the Water Authority Veluwe. These data were collected and analyzed by the Staring Centre (the research institute of the Ministry of Agriculture and Fisheries). The water flows and concentrations of the other streams were calculated by interpolation of this data, according to ICIM (1994). From 1978 onwards, the water quality in Lake Nuldernauw and Lake Wolderwijd was measured 12 times a year by the Dutch Water Authority Directorate IJsselmeer Area. Data on evaporation and precipitation were obtained from daily measurements by the Royal Dutch Meteorological Institute. Sampling was done at fixed locations. Standard methods for sampling and laboratory analysis were used. In this paper, we present data from the period 1985 - 1994 (10 years).

RESULTS AND DISCUSSION

Water flows by streams

Figure 2 shows that the water flow of the streams is strongly influenced by the net rainfall (precipitation minus evaporation). The relation between these parameters is exponential. As discussed in Negate *et al.* (1997) the streams have a short reaction time to precipitation. The evaporation reflects the storage of water into groundwater, since the majority of the catchment area has a high groundwater level (Breeuwsma *et al.*, 1989). In summer, the net rainfall is negative. The groundwater level is low because groundwater is used for agricultural irrigation (Van der Molen, 1994). In winter, a surplus of precipitation, combined with a high groundwater level results in a high discharge by the streams. In this period, the water flow totals 80% of the annual water flow (De Boer *et al.*, 1996).

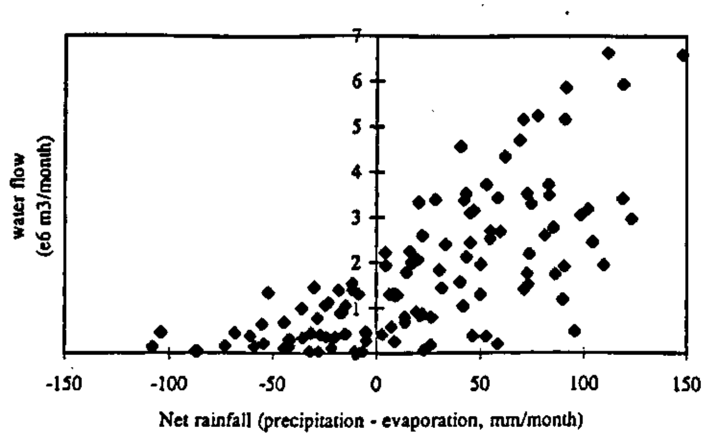


Fig. 2 Monthly net rainfall and water flows of the streams

Phosphorus loads by streams

In figure 3, the relation between water flow and phosphorus in the stream Schuitenbeek is presented.

Figure 3a shows water flow and phosphorus concentration. As shown in figure 2, water flows are mainly affected by net precipitation. With increasing water flows and high groundwater tables, the phosphorus concentration in streams increases (Negate *et al.*, 1997). This is a result of increased surface runoff and subsurface runoff adding an extra phosphorus load from the phosphorus saturated catchment area (De Boer *et al.*, 1996). High water flow, causing limited hydraulic residence time in streams, results in low biological activity in the surface water. In winter, a period with 80% of the annual water flow, the phosphorus concentration has an average of 0.47 mg P l^{-1}. In summer, the average phosphorus concentration is 0.39 mg P l^{-1} (De Boer *et al.*, 1996).

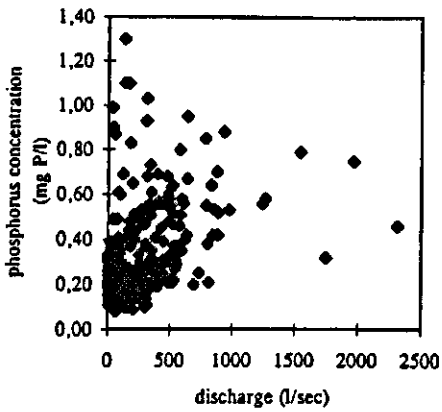


Fig. 3a Water flows and phosphorus concentration by stream Schuitenbeek

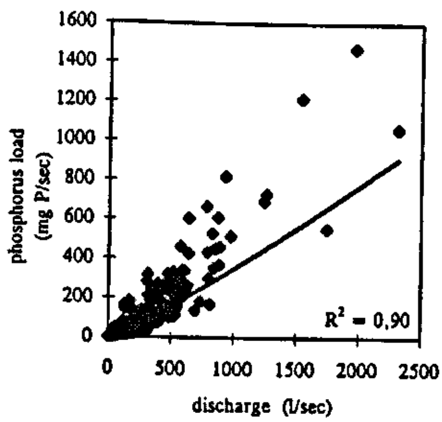


Fig.3b Discharges of water and phosphorus by stream Schuitenbeek

Figure 3b shows water flow and phosphorus load. The high water flow by streams in winter lead to a phosphorus load. That is higher than proportional because of more water containing higher phosphorus concentrations (De Boer *et al.*, 1996; Negate *et al.*, 1997).

Phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd

In figure 4, the relation between phosphorus loads in streams and phosphorus concentration in the lakes is presented. Figure 4a shows the loads of streams and concentration in Lake Nuldernauw, figure 4b shows the loads of streams and concentration in Lake Wolderwijd. The loads by the streams greatly vary monthly. This variation explains the erratic course of the phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd. Compared to Lake Wolderwijd, Lake Nuldernauw has a smaller volume of water and a smaller hydraulic residence time. Therefore, in Lake Nuldernauw the effects are more extreme (Van Ballegooijen and Van der Molen, 1994). Compared to Lake Wolderwijd the phosphorus concentrations in Lake Nuldernauw are higher because of the higher loads.

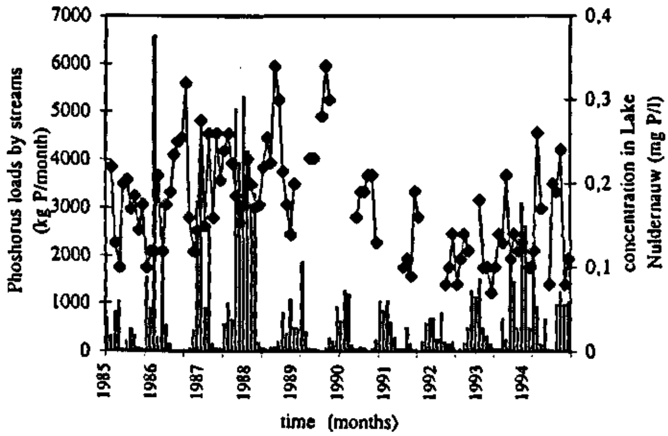


Fig. 4a Phosphorus loads by streams and phosphorus concentration in Lake Nuldernauw

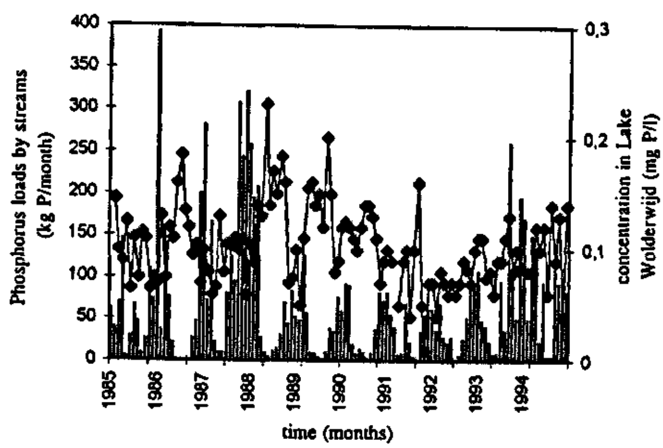


Fig. 4b Phosphorus loads by streams and phosphorus concentration in Lake Wolderwijd

Data from the period 1985-1988 generally show lower than average phosphorus concentration in Lake Wolderwijd and Lake Nuldernauw. In this period, a lowering of the groundwater level was seen (Meijer and Hosper, 1995). This resulted in decreased runoff and about 40% lower phosphorus concentrations in the streams (Van der Molen and Van Ballegooijen, 1993) diluting the loads. Data from the year 1992 also show a lower phosphorus concentration. This is a result of large scale biomanipulation in 1991 (Meijer and Hosper, 1995) and intensified lake flushing (Van Ballegooijen and Van der Molen, 1994). From 1992 onwards, data generally show higher phosphorus concentrations in Lake Wolderwijd and Lake Nuldernauw. In this period, the groundwater level increased (Meijer and Hosper, 1995), resulting to increased runoff and higher phosphorus concentrations in the streams (Van der Molen and Van Ballegooijen, 1993).

Tested for the relation between water flows, phosphorus concentration or phosphorus loads added by the streams in winter and the phosphorus concentration in Lake Nuldernauw and Lake Wolderwijd in spring and summer were done for the period January - March and April - September, respectively. For Lake Nuldernauw, the best correlation was between phosphorus loads by streams in March and the average phosphorus concentration in April and May (R^2 = 0.74) (figure 5a). For Lake Wolderwijd, the best correlation was between phosphorus loads in the streams in February and March and the average phosphorus concentration in April - June (R^2 = 0.65) (figure 5b).

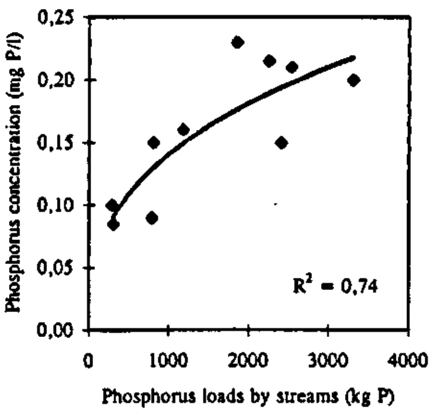


Fig. 5a Phosphorus loads by streams (March) and phosphorus concentration in Lake Nuldernauw (April - May)

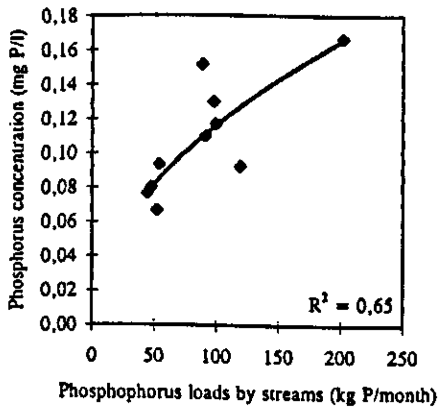


Fig. 5b Phosphorus loads by streams (February - March) and phosphorus concentration in Lake Wolderwijd (April - June)

The period, in which the relationship is valid, is for Lake Nuldernauw smaller (1 month) than for Lake Wolderwijd (3 months). This can be explained by the hydraulic residence time of both lakes (table 1).

Both curves (figure 5a and 5b) have a plane shape. At the beginning of the curve, the phosphorus concentration in the lakes increases relatively fast with increasing phosphorus load. Increased loads mainly appear with an increased water flow (figure 3b). The increasing water flows of streams have a higher phosphorus concentration (figure 3a). Therefore, the concentration in the receiving surface water will increase. On the other hand, at a higher water flow the phosphorus in the lakes will be washed out due to the short hydraulic residence time.

Compared to Lake Wolderwijd, the phosphorus concentration in Lake Nuldernauw has a higher correlation with the loads carried by streams. This is because streams have a greater influence in the P budget to Lake Nuldernauw, than of Lake Wolderwijd (table 2). The water quality of Lake Wolderwijd and Lake Nuldernauw is at least partly determined by the hydrology. The differences between yearly data are due to the difference between precipitation and evaporation in late winter. The relation between phosphorus concentration in lake water in spring and net precipitation in winter has been demonstrated for other lake systems in the Netherlands as well (*eg* Lake Botshol (Ouboter, 1997)).

The phosphorus loads of streams in late winter are regulating the phosphorus concentration in spring in Lake Nuldernauw and Lake Wolderwijd. The concentration in spring is of biological importance, since it is the beginning of the growing season. In this period, competition starts between algae and macrophytes. A high phosphorus concentration results in high chlorophyll-a concentration and high turbidity (Reeders *et al.*, 1997). This represses the growth of macrophytes in the lakes. Recent developments show that macrophytes recover at increased water transparencies and low phosphorus concentrations (Coops *et al.*, 1997; Van der Molen & Boers, 1997). A large amount of precipitation in the late winter, resulting in high phosphorus concentration in the lakes in spring, may hence negatively affect the ecosystem in the growing season to come.

CONCLUSIONS

- The water flow of the streams draining to Lake Wolderwijd and Nuldernauw is strongly influenced by the net rainfall.
- High water flows in streams mainly appear in winter and lead to more than proportional high phosphorus loads, because of a higher concentration. The streams are mainly loaded by agricultural runoff.
- The phosphorus loads by streams in the late winter (February - March) are regulating the phosphorus concentration in spring (April - June) in Lake Nuldernauw and Lake Wolderwijd. This implicates that the net precipitation in late winter affects the ecosystem in the growing season.

ACKNOWLEDGEMENT

A huge number of people have been sampling and doing chemical measurements to fill the database since 1985. We acknowledge their effort and dedication through which we were enabled to make the analysis. Furthermore, the authors wish to thank Mr Van Kempen of the Water Authority Veluwe for the delivery of data and Ms Beaton for correcting the English translation.

REFERENCES

Breeuwsma, A., J.G.A. Reijerink, O.F. Schoumans, D.J. Bus and H. van het Loo (1989). *Phosphorus loads to soil, ground and surface water in the catchment area of the stream Schuitenbeek.* DLO-Staring Centrum rapport 10, Wageningen (in Dutch).

Coops, H., R.W. Doef, B.J. de Witte and M.S. van den Berg (1997). Recovery of aquatic vegetation in the Lake IJsselmeer region. *De Levende Natuur* **98** (1), 8 - 13 (in Dutch, with English summary).

De Boer, W.A., G.L. Negate, H.P. Oosterom and C.W.J. Roest (1996). *Discharges of water, nitrogen and phosphorus in the Schuitenbeek-area. Discharges of streams 1988-1994.* DLO-Staring Centrum rapport 424.2, Wageningen (in Dutch).

ICIM (1994). *Basis data balance-calculations Lakes Randmeren. Collecting data, methods and results.* ICIM rapport 1994-547, Rijswijk (in Dutch).

Meijer, M.-L. and S.H. Hosper (eds.) (1995). *Restoration of Lake Wolderwijd, The Netherlands, evaluation of a drastic fish stock reduction.* RIZA notanr 94.058, Lelystad (in Dutch, with English summary).

Negate, G.L., H.P. Oosterom, J. Pankow, J. Oude Voshaar, C.W.J. Roest and A. van der Toorn (1997). *Discharges of water, nitrogen and phosphorus in the Schuitenbeek-area. Field research 1988-1994.* DLO-Staring Centrum rapport 424.1, Wageningen (in Dutch).

Ouboter, M.R.L. (1997). *Dynamics eutrophication Botshol.* Waterloopkundig Laboratorium, Delft (in Dutch).

Projectgroep Eutrofiëringsonderzoek Randmeren (PER) (1982). *Eutrophication research Lake Wolderwijd-Nuldernauw in the period May 1976 till September 1979.* Lelystad (in Dutch).

Reeders, H.H. and T.H. Helmerhorst (1996). *On the way to clearity. A reorientation of BOVAR towards the year 2000.* Directie IJsselmeergebied, Rapport nr 96.01, Lelystad (in Dutch).

Reeders, H.H., P. Boers, D.T. van der Molen and T.H. Helmerhorst (1997). Cyanobacterial dominance in Lake Veluwemeer and Lake Wolderwijd, the Netherlands. *This issue*

Van Ballegooijen, L. and D.T. Van der Molen (1994). *Budgets for water and nutrients for the lakes between the sluice Roggebotsluis till the sluice Nijkerkersluis in the period 1976 - 1992.* RIZA werkdocument 94.082X, Lelystad (in Dutch).

Van der Molen, D.T. (1994). *Recent developments in the discharges and loads by nutrients of streams in the western edge of the Veluwe.* RIZA werkdocument 94.107X, Lelystad (in Dutch).

Van der Molen, D. and L. van Ballegooijen (1993). *Budgets for water and mass for Lake Wolderwijd and Lake Nuldernauw in the period 1976 - 1991.* RIZA werkdocument 93.102X, Lelystad (in Dutch).

Van der Molen, D.T. and P.C.M. Boers (1997). Eutrophication and oligothrophication of Lake Veluwemeer. *De Levende Natuur* 98 (1), 2 - 7 (in Dutch, with English summary).

PROCESSES

OXYGEN DISTRIBUTION AND CONSUMPTION IN THE SEDIMENTS OF TEMPERATE LAKES

Jean-Pierre R.A. Sweerts[1], Eric Loopik and Thomas E. Cappenberg[2]

[1]*Delft Hydraulics, P.O. Box 177, 2600 MH DELFT*
[2]*Center for Estuarine and Marine Ecology, P.O. Box 140, 4400 AC Yerseke*

ABSTRACT

Distribution of oxygen into the sediments of a variety of temperate lakes was measured with microelectrodes. The lakes varied in maximum depth from 2-50 m, in surface area from 5-34700 ha, and in anual primary production from 20-500 $gC.m^{-2}.d^{-1}$. Oxygen consumption and the depth of oxygen penetration in epilimnetic sediments were only slightly related to these three lake characteristics. Oxygen penetration depths in summer were within a small range (1.2-6.5 mm) in all sandy, gyttja, and flocculant sediments. Only in shallow sandy shore sediments oxygen penetrated up to 15 mm. In photosynthetic active sediments of oligotrofic lakes were fluctuations in oxygen penetration depth observed between 0 and 3 mm during the day. Oxygen was present up to 20 mm in the littoral sediments of an oligotrofic lake due to transport of oxygen into the sediments by the roots of plants (Carex spp.). Seasonal variations in oxygen penetration depth and consumption were present in oligomesotrofic and eutrophic lakes ($1.6 < Q_{10} < 4.9$).

Oxygen penetration depth (0.1-12.2 mm) and consumption (5.7-44.6 $mmol.m^{-2}.d^{-1}$) of profundal sediments were positively and inversly related to the depth of the lake, respectively. Oxygen penetration depth at 10 m depth varied from 0.1 mm in a meso-eutrofic lake to 4.8 mm in an oligotrophic lake of similar morphometry.

INTRODUCTION

Until recently, it was generally believed that oxygen penetrated up to several cm into the surface sediments of aquatic systems. This idea originated from the observation that the surface layer of sediments was often reddish-brown coloured. The introduction of oxygen microelectrodes to aquatic ecology has enabled the actual measurement of fine-scale oxygen gradients in sediments (Revsbech et al. 1980[a+b]). With the exception of deep sea sediments, it has been found that oxygen penetrates to a depth of only a few mm into most sediments. To date, most of these measurements have been made in the marine environment, either in microbial mats (Revsbech et al. 1983, Jorgensen et al. 1988), tidal sediments (Andersen and Helder 1987), coastal sediments (Revsbech et al. 1980[b], Lindeboom et al. 1985, Silverberg et al. 1987), or in the deep sea (Reimers et al. 1984; 1987, Reimers and

Smith 1986). Recently, sediment-oxygen profiles have also been obtained from a few small freshwater lakes (Sweerts et al. 1986, 1989, Carlton and Wetzel 1987) from lake Washington (Kuivela et al. 1988) and from Lake Superior of the Laurentian Great Lakes (Carlton et al.1989, Richardson and Nealson, in press).

In most sediments, the depth of oxygen penetration (SOP) is related to its consumption (Revsbech et al. 1980*). Sediment oxygen consumption (SOD) is generally used as a measure for the aerobic mineralization of organic material in aquatic sediments not inhabitad by invertebrates. Our study, which examines oxygen distribution in the epilimnetic and hypolimnetic sediments of a variety of Canadian, Dutch, and Danish lakes, provides further evidence that oxygen is rapidly consumed as it diffuses into lake sediments. We determined SOP and SOD of sediments for lakes that varied widley in terms of depth (2-50m), surface area (5-35000 ha), and lake primary productivity (20-500 $gC.m^{-2}.day^{-1}$). Trophic state and depth were important determinants of SOP and SOD in profundal sediments. For epilimnetic sediments these relations were not as pronounced. The importance for SOP and SOD of temperature, porosity, the availebility of organic material and light are discussed.

GENERAL CONCLUSIONS

Since the introduction of oxygen microelectrodes in aquatic ecology (Revsbech et al. 1980), oxygen penetration depths have been measured in the sediments of various marine and freshwater systems. The measured oxygen penetration depths of several studies are summarized in Figure 8. The range of oxygen penetration depth in our study is comparable with the range measured in coastal marine sediments (Revsbech et al. 1980, Silverberg et al. 1987). Similar to our observations in lake sediments (Table 1), sandy near shore marine sediments had oxygen penetration depths up to > 10 mm (Andersen et al. 1987). The oxygen penetration depth at 40 m depth in Trout lake (12.2 mm) is within the range of both the marine coastal sediments and the sediments of lake Superior. All SOP measured in the deep sea were higher (Reimers et al. 1987). The measured oxygen consumption in the top 0.1 mm of mats of Beggiatoa spp. covering the sediments is comparable with mats of Beggiatoa spp. in the marine environment (Nelson et al. 1986). The most active marine sediments are the microbial mats with respect to photosynthesis and oxygen consumption (Revsbech et al. 1983). Comparable mats like the microbial mats in the marine environment were not found in our study of freshwater lakes. In the marine microbial mats, the sulfur cycle is of general importance, while much lower sulphur (sulfate) concentrations exists in the freshwater environment.

In general, oxygen distribution in the freshwater lakes were in the same range as coastal marine sediments. However, some different bacterial populations and biogeochemical reactions are involved in the oxygen consumption in freshwater and marine sediments (Sweerts et al. 1991B).

PHOSPHORUS-BINDING FORMS IN THE SEDIMENT OF AN OLIGOTROPHIC AND AN EUTROPHIC HARDWATER LAKE OF THE BALTIC LAKE DISTRICT (GERMANY)

Thomas Gonsiorczyk, Peter Casper and Rainer Koschel

Institute of Freshwater Ecology and Inland Fisheries, Department of Limnology of Stratified Lakes, Alte Fischerhuette 2, D-16775 Neuglobsow, Germany

ABSTRACT

The vertical distribution of various phosphorus (P)-binding forms, associated potential P-binding partners and the composition of dry material were investigated in the bottom sediments of the dimictic oligotrophic Lake Stechlin and the dimictic eutrophic Lake Feldberger Haussee. Reductant soluble P (Fe- and Mn-bound) at the sediment surface (0-1 cm) was considerably higher in the oligotrophic Lake Stechlin (1.29 g kg^{-1}) than in the eutrophic Lake Haussee (0.32 g kg^{-1}). The amounts of dissolved, loosely adsorbed, metal oxide- and calcium carbonate bound P were higher in the eutrophic lake. The depth profiles of the investigated P species indicated that the mobilization of Fe- and Mn-bound P is the most important mechanism of P-release in oligotrophic lakes, whereas the mobilization of recently sedimented labile organic bound P seems to be the driving force of P-release in eutrophic lakes. In both lakes autochthonous calcite precipitations occurs during the summer months. The coprecipitation of P with calcite is an important self-cleaning mechanism in eutrophic hardwater lakes and contributes to the permanent burial of P in the sediments. Although, the precipitation of calcite is inhibited by the presence of high concentrations of soluble reactive P, the coprecipitation of P with calcite seems to be enhanced.

KEYWORDS

calcium carbonate; eutrophication; lakes; phosphorus fractionation; sediments; trophic status

INTRODUCTION

The chemical composition of lake sediments, particularly the content of total phosphorus (TP), is often related to the trophic state (Rybak, 1969; Flannery et al., 1982; Mothes, 1980) or to the trophic development of lakes (Niessen and Sturm, 1987; Schneider et al., 1990; Gonsiorczyk et al., 1995). On the other hand, the TP content of the sediments is as well often mentioned to explain the extend of P-release by sediments (Nürnberg, 1988; Ostrowsky, 1989; Sas, 1989) and dredging of the top P-rich sediment layers is often considered for lake restoration (Kleeberg and Kozerski, 1997). The P-content of sediments is the result of the P-binding capacity, which depends on the sediment composition, sedimentation rate, chemical conditions and the extent of diagenetic processes in the sediments (Boström et al., 1982; 1988; Hupfer et al., 1995). Trophic status seemed to be a secondary factor in controlling the fractional composition of P in the sediments of 43 Spanish reservoirs and regional variations seemed to be more important (Lopez and Morgui, 1993). In the calcium- and carbonate rich lakes of northern Germany the coprecipitation of P during pelagic calcite ($CaCO_3$) precipitations is expected to have a great influence on the P-binding capacity of the sediments

(Koschel et al., 1987; Dittrich et al., 1997). Accordingly, this process must have an affect on the fractional composition of P in the sediments.

In this study we investigated the vertical distribution of various P-species in the bottom sediments of the oligotrophic Lake Stechlin and the eutrophic Lake Feldberger Haussee in relation to the contents of $CaCO_3$ and organic matter (OM). In both lakes calcite precipitations occur during the summer months (Koschel, 1990; 1995). We also compared the fractional composition of P in the bottom sediments to assess the stability and mobility of various P-binding forms and examined to what extent the composition of particulate material and the chemical conditions affect the pattern of P-binding forms.

METHODS

Site description

Lake Stechlin is one of the last oligotrophic lakes in the North-German lowlands (Casper, 1985; Mietz et al., 1994). The lake is characterized by always prevailing oxic conditions in the whole water body (Koschel, 1995). By way of contrast, Lake Feldberger Haussee provides a classic example of eutrophication history of many hardwater lakes in the Baltic Lake District (Krienitz et al., 1996). The lake was originally slightly eutrophic and was heavily polluted by sewage inlet. The annual mean concentrations of soluble reactive P (SRP) between 1978-1985 were in excess of 1 mg l^{-1} (Koschel et al., 1990; Koschel, 1995). As a result of sewage diversion from the catchment since 1980 and a biomanipulation experiment since 1985 (Kasprzak et al., 1988) the water quality gradually improved but annual TP-concentrations are still about 0.1 mg l^{-1}. The hypolimnion regularly deoxygenates and diffusion of SRP out of the sediments leads to high accumulation of SRP during summer stagnation (Tab. 1).

Table 1: Morphometric and trophic characteristics of Lake Stechlin and Lake Feldberger Haussee (Casper and Koschel, 1995; Koschel, 1995).

		Lake Stechlin	Lake Haussee
Area	[km^2]	4.25	1.30
max. depth	[m]	68.5	12.0
SRP / TP (spring circulation 1995)	[mg l^{-1}]	0.003 / 0.024	0.099 / 0.178
SRP / TP (hypolimnion) [a]	[mg l^{-1}]	0.025 / 0.037	0.741 / 0.824
SRP / TP (interstitial water, 0-10 cm) [a]	[mg l^{-1}]	0.521 / 0.592	2.945 / 3.046
P-release rate by sediments [b]	[mg m^{-2} d^{-1}]	0.35	3.04

[a] end of summer stagnation 1995 [b] Mean diffusion rate (1995) calculated from the SRP gradients at the sediment water interface (Gonsiorczyk et al., in press)

Sampling and chemical analysis

Sediment cores were taken at the deepest part of the south-west-basin of Lake Stechlin (water depth = 32 m) and at the deepest part of the north-basin of Lake Haussee (water depth = 8.5 m) using a Jenkin-sampler. The cores were immediately sectioned following the method of Casper (1992). In Lake Stechlin the P-binding forms were analyzed in March, April, May and August 1995. Mean values of these investigations were considered for the evaluation. In Lake Haussee the P-fractionation was done during spring circulation in April 1997. The P-fractions were determined using a sequential extraction procedure according to Psenner et al. (1984) and modified by Hupfer et al. (1995). A schematic representation of the procedure is given in table 2. In the most samples the sum of the sequential extracted P was in good agreement with the independently determined TP with an mean extraction efficiency of 90±12 %. Fractionation results are presented on the basis of total extracted P, with the sum of all fractions totaling 100 %.

Table 2: Sequential extraction procedure of P and potential P-binding elements and expected P-species in the fractions (SRP = soluble reactive P, NRP = non reactive P).

Extractants (time)		Expected P-forms
1. 1 M NH_4Cl (0.5 h)	SRP	dissolved SRP and loosely adsorbed P
	NRP	dissolved (<0,45 µm) and NH_4Cl soluble NRP
2. 0.11 M $Na_2S_2O_6$ (bicarbonate/dithionite, BD) (1 h)	SRP	redox-sensitive P mainly bound to Fe-hydroxides and Mn-compounds
	NRP	redox-sensitive organic P
3. 1M NaOH (16 h)	SRP	P bound to metal oxides mainly of Al and Fe, inorganic P-compounds soluble in bases
	NRP	P in microorganisms including poly-P, organic P in detritus, P bound to humic compounds
4. 0.5 M HCl (16 h)	SRP	P bound to carbonates and apatite-P, traces of hydrolyzed organic P
	NRP	HCl soluble organic P
5. $K_2S_2O_8$ (30 min, 134°C)	TP	organic and other refractory P (residual P)

The interstitial water was obtained by centrifugation and a subsequent filtration through cellulose-nitrate-filters (0.45 µm). As well the extracts of P-fractionation were centrifuged and filtered (using polyamid-filters for the NaOH-extracts). The SRP-concentrations were analyzed photometrically by the molybdenum blue method (Murphy and Riley, 1962). TP was transformed into SRP by adding $K_2S_2O_8$ (30 min at 134°C). The difference between TP and SRP, so called non reactive P (NRP), is assumed to be organically bound. The concentrations of the most important potential P-binding elements (Fe, Al, Mn and Ca) were measured in the NH_4Cl-, BD-, NaOH- and HCl-extracts with flame atomic absorption spectrometry (AAS) and ICP/AES (inductive coupled plasma/atomic emission spectrometry). The content of organic matter and $CaCO_3$ were determined as loss on ignition at 550°C and 900°C, respectively (Mothes, 1980).

RESULTS AND DISCUSSION

General sediment characteristics

Although, the trophic state of both lakes is completely different the composition of particulate matter in the top 8 cm of the sediments, particularly the content of organic matter, is relatively similar (Fig. 1). However, the SRP-concentrations in the interstitial water are much higher in the sediment of Lake Haussee.

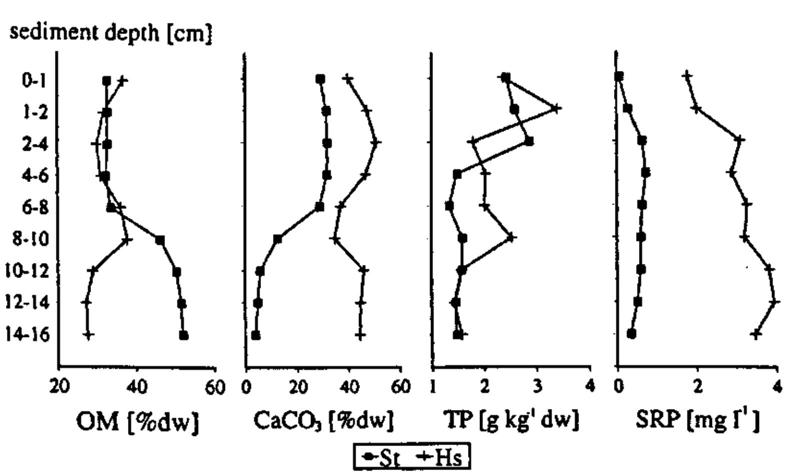


Figure 1: Sediment composition in Lake Stechlin (St) and Lake Feldberger Haussee (Hs) (dw = dry weight)

According to the results of Cs-dating (Casper, 1994) the changes of the calcite content in the sediments of both lakes were related to the trophic development (Gonsiorczyk et al., 1995). As a result of a strong intensification of calcite precipitation in Lake Stechlin at the beginning of the 20th century the calcite content increases at a sediment depth of 8 cm (Fig. 1). Comparable gradients of the calcite content were found in the sediments of many other lakes and were attributed to an increased calcite precipitation due to an increase of primary production by nutrient enrichment (Niessen and Sturm, 1987; Schneider et al., 1990; Gonsiorczyk et al., 1995). On the other hand, in Lake Haussee the intensity of calcite precipitation was inhibited during the time when the lake was highly eutrophic. The calcite content is reaching a minimum in the sediment layers between 6-10 cm that had been deposited during the maximum of eutrophication (Krienitz et al., 1996; Koschel, 1995). In highly eutrophic lakes calcite precipitations may be inhibited due to high SRP concentrations (Kleiner, 1988) and/or marked short-term fluctuations of H_2CO_3 (Koschel et al., 1987).

Adsorbed and dissolved Phosphorus (NH_4Cl-P)

Almost all of adsorbed P was inorganically bound. In both lakes a large fraction of NH_4Cl-soluble P was present as dissolved P in the interstitial water (29 ± 12 %, range: 6-48 %). Although, the concentrations of dissolved P were increasing with sediment depth (Fig. 1), the portion of NH_4Cl-soluble P was decreasing with depth (Fig. 2). This may be due to desorption of P in the deeper sediment layers where the pH is decreasing (Gonsiorczyk et al., in press). Some of the loosely adsorbed P may probably associated to $CaCO_3$ crystals (Psenner and Puscko, 1988). The NH_4Cl-extraction solutes relatively large amounts of Ca (Fig. 3). This indicates that a large fraction of adsorbed P was previously associated with calcite particles. The portion of loosely adsorbed P (corrected for dissolved P) was higher in the sediment of the eutrophic Lake Haussee (4-8 % of TP) than in the oligotrophic Lake Stechlin (1-3 % of TP).

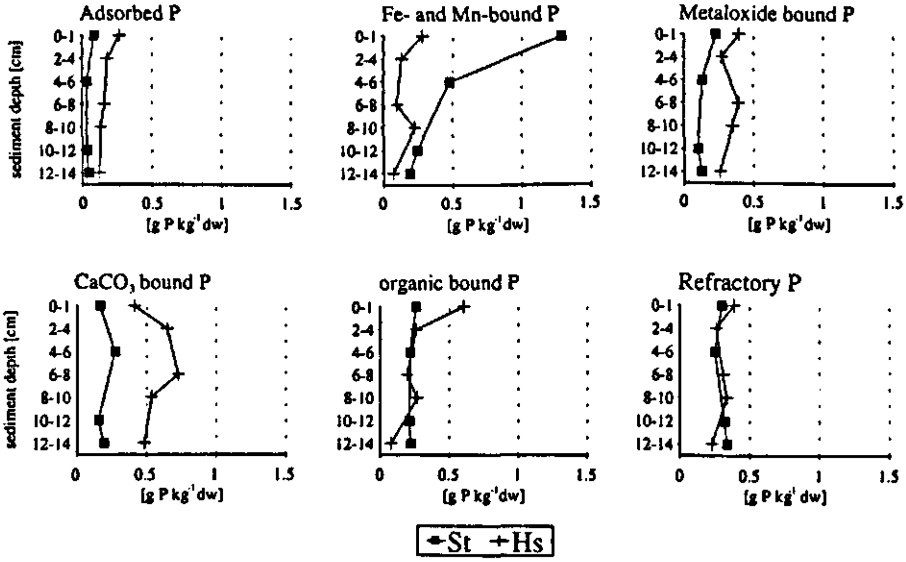


Figure 2: Sequential extracted P compounds in different sediment depths of Lake Stechlin (St) and Lake Feldberger Haussee (Hs)

Reductant soluble Phosphorus (BD-P)

In both lakes most of the reductant soluble P was extracted as SRP and therefore is assumed to be bound to Fe-hydroxides and Mn-compounds (Tab. 2). In the oligotrophic Lake Stechlin the amount of Fe- and Mn-bound P was considerable higher than in Lake Haussee (Fig. 2). This is also represented by higher amounts of extracted Fe and Mn (Fig. 3). The maxima of reductant soluble P, Fe and Mn were found at the sediment surface (Fig. 2 and 3). These maxima may reflect the composition of recently sedimented material (Hupfer et al., 1995). They also may be the product of an oxic sediment surface that acts as a boundary layer for upwards diffusing P, Fe and Mn (Farmer et al., 1994). In the top 5-10 mm of the bottom sediment of Lake Stechlin oxic conditions are always present (Sass et al., 1997). Fe- and Mn-bound P was the most important fraction of TP (54 %) in the top centimeter of the sediment (1.29 g kg^{-1}). Its amount was decreasing with depth to values of about 0.2 g kg^{-1} (> 10 cm). The P-release due to the burial of the top oxic sediment layer into deeper anoxic zones seems to be the main reason for P-release by the sediment in Lake Stechlin (Gonsiorczyk et al., in preparation). In the eutrophic lake, Fe- and Mn-bound P was considerable lower (0-1 cm: 0.32 g kg^{-1}). The decrease of Fe- and Mn-bound P was not as pronounced. Surprisingly, the amount of Fe- and Mn-bound P was increasing at a sediment depth of 8-10 cm and corresponds with a large increase of reductant soluble Fe. This second maximum of reductant soluble P and Fe seems to be connected with the maximum of eutrophication in Lake Haussee.

Metal oxide bound phosphorus (NaOH-SRP)

The amounts of P bound to Fe- and Al-oxides were higher in the eutrophic lake (Fig. 2). This corresponds with higher amounts of extracted Al in the sediment of Lake Haussee (Fig. 3). Most of total extracted Al was soluble by NaOH extraction. In the sediment of Lake Feldberger Haussee about 83.0±1.4 % of total extracted Al were dissolved by NaOH-extraction, in Lake Stechlin only 57.9±7.6 %. In both lakes the maxima of metal oxide bound P were found at the sediment surface (0-1 cm). This was also observed in the sediments of many other poly- and dimictic eutrophic lakes (Hupfer et al., 1995; Farmer et al., 1994; Penn et al., 1995; Ramm and Scheps, 1996). In the sediment of Lake Haussee a second maximum of Fe- and Al-oxide bound P was found in deeper sediment layers (6-8 cm).

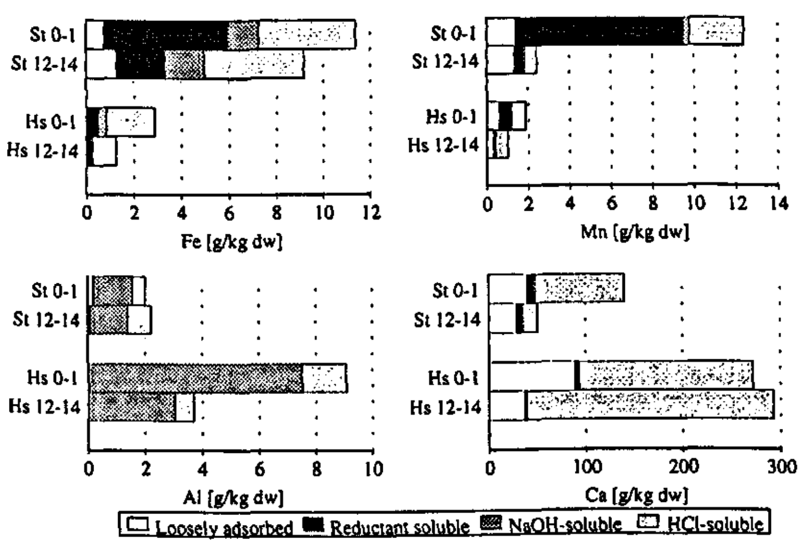


Figure 3: Sequential extracted amounts of Fe, Mn, Al and Ca in two selected sediment layers of Lake Stechlin and Lake Feldberger Haussee

$CaCO_3$ bound phosphorus (HCl-P)

Most of HCl-soluble P was inorganically bound. Additionally, most of total extracted Ca was soluble by HCl-extraction (Fig. 3). Since autochthonous calcite precipitation controls the $CaCO_3$ content of the sediments in the investigated lakes (Proft, 1984), most of HCl-soluble P is in all probability coprecipitated with autochthonous precipitated $CaCO_3$. Although, the $CaCO_3$ content in the top 8 cm of the sediment in Lake Stechlin is only a little bit lower than in Lake Haussee (Fig. 1) the amount of $CaCO_3$ bound P was much higher in the eutrophic lake (Fig. 2). In the sediment of Lake Haussee on average about 32 % of TP are associated with $CaCO_3$ whereas in Lake Stechlin only about 14 % (0-14 cm). Similar portions of $CaCO_3$-bound P as in Lake Haussee were found in the calcareous sediments of the dimictic, meso- to eutrophic Lake Breiter Luzin, the mesotrophic Lake Tegernsee (Hupfer, 1995) and in the polymictic, highly eutrophic Lake Blankensee (Ramm and Scheps, 1996). Calcium-bound P is a relatively stable fraction of sedimentary P and contributes to a permanent burial of P in the sediments (Hupfer et al., 1995).

Although, the $CaCO_3$ content in the bottom sediment of Lake Stechlin is drastically increasing (Fig. 1) this did not lead to an appropriate increase of $CaCO_3$-bound P. Due to the low SRP-concentrations in the pelagial of Lake Stechlin (Tab. 1), coprecipitation of SRP with $CaCO_3$ may not be worth mentioning. In Lake Haussee the maximum amount of $CaCO_3$-bound P was found in those sediment layers where the $CaCO_3$ content reached a minimum (Fig. 1 and 3). These sediment layers (6-10 cm) were deposited during the periods of highest eutrophication. The ratios of SRP/Ca in the HCl-SRP extracts were higher in the sediment layers between 6-10 cm (3.64±0.11 mg P/ g Ca) than in the other investigated layers (2.00±0.12 mg P/ g Ca). Consequently, the coprecipitation of P in Lake Haussee was more intensive during the maximum of eutrophication. The nucleation and growth of calcite crystals are inhibited by the presence of high SRP-concentrations (Kleiner, 1988). This leads to higher $CaCO_3$-oversaturations before calcite is precipitated. The intensified coprecipitation of P during the maximum of eutrophication in Lake Haussee may be the result of higher $CaCO_3$-oversaturations and/or higher SRP-concentrations in the pelagial. Similar ratios for the coprecipitation of P with calcite as in Lake Haussee were calculated for other lakes with completely different methods: i.e. Rossknecht (1980): 0.1-1.0 %; Kleiner (1988): 0.34 ± 0.20 %; Jäger and Röhrs (1990): 0.1-0.5 %; Dittrich et al. (1997): 0.1 %.

Labile organic (NaOH-NRP) and refractory bound phosphorus (residue P)

In the eutrophic lake the amount of labile organic P was considerable decreasing with sediment depth (Fig. 2). This can be attributed to the microbial mineralization of recently sedimented organic material (see also Fig. 1). Similar observations were also made in many other lakes (Farmer et al., 1994; Hupfer et al., 1995; Penn et al., 1995). Since methanotrophic and other bacteria contribute to the P uptake in oxic sediment layers (Sinke et al., 1992; Gächter et al., 1993) it was expected to find a similar surface enrichment of labile organic P in the sediment of Lake Stechlin. However, the amount of labile organic P shows only an unimportant maximum at the sediment surface (0-1 cm) and remains relatively constant with increasing sediment depth. The organic P-pool may not be exposed to intensive diagenetic changes while the quality of organic matter in the sediment of Lake Stechlin is expected to be mostly refractory (Casper, 1992). In oligotrophic lakes, a larger fraction of P settling with organic detritus is converted to refractory organic compounds by benthic microorganisms than in eutrophic lakes (Gächter et al., 1993). However, the amounts of refractory organic and other bound P were relatively similar in both lakes and remained relatively constant in all sediment depths (Fig. 2). Nevertheless, in the deeper sediment layers (10-14 cm) of Lake Stechlin the portion of refractory P on TP (30 %) was greater than in the eutrophic lake (18 %). Like $CaCO_3$ bound P, this fraction is not exposed to intensive diagenetic processes and contributes to the permanent burial of P in the sediment.

CONCLUSIONS

In contrast, to the fractional composition of P the content of total P in the sediments of the investigated lakes is not directly dependent from the trophic state. In eutrophic lakes, the coprecipitation of P during autochthonous calcite precipitations increases the P-binding capacity of the sediments. Therefore, technologies that increase the extent of calcite precipitations during periods of high P-concentrations in the pelagial could be an effective tool to enhance the net sedimentation of P in eutrophic hardwater lakes. In eutrophic lakes, the mineralization of organic matter seems to be the driving force of P-release by sediments. In oligotrophic lakes, the mobilization of Fe- and Mn-bound P due to the burial of the top oxic sediment layers into deeper anoxic zones seems to be the main mechanism of P-release. To prevent eutrophication of these lakes it is imperative to maintain oxic conditions at the top 5-10 mm of the sediments.

ACKNOWLEDGMENTS

We acknowledge Hans Jürgen Exner (Institute of Freshwater Ecology and Inland Fisheries, AAS-measurements) and Volker Scheps (Landesamt für Geowissenschaften und Rohstoffe, Kleinmachnow, ICP/AES-measurements) for the analysis of potential P-binding elements and Graham Hall and Dominik Hepperle for their comments on the manuscript.

REFERENCES

Boström, B., Jansson, M. and Forsberg, C. (1982). Phosphorus release from lake sediments. *Arch. Hydrobiol. Beih. Ergebn. Limnol.*, **18**, 5-59.

Boström, B., Andersen, J. M., Fleischer, S. Jansson, M. (1988). Exchange of phosphorus across the sediment-water interface. *Hydrobiologia*, **170**, 229-244.

Casper, P. (1992). Methane production in lakes of different trophic state. *Arch. Hydrobiol. Beih. Ergebn. Limnol.*, **37**, 149-154.

Casper, P. (1994). Die Cäsiumdatierung von Sedimenten unterschiedlicher mikrobieller Aktivität. Deutsche Gesellschaft für Limnologie, Jahrestagung 1994. Hamburg, *Erweiterte Zusammenfassungen*, 386-389.

Casper, P. and Koschel, R. (1995). Description of Lake Stechlin. *Limnologica*, **25**, 281-284.

Casper, S. J. (ed.) (1985). Lake Stechlin- A temperate oligotrophic lake. *Monogr. Biol. Vol.*, **58**. Dr. W. Junk Publ., Dordrecht-Boston-Lancaster.

Dittrich, M., Dittrich, T., Sieber, I. and Koschel, R. (1997). A balance analysis of phosphorus elimination by artifical calcite precipitation in a stratified hardwater lake. *Wat. Res.*, **31** (2), 237-248.

Farmer, J. G., Bailey-Watts, A. E. Kirika, A. and Scott, C. (1994). Phosphorus fractionation and mobility in Loch Leven sediments. *Aquat. Conserv.: Mar. Freshwat. Ecosyst.*, **4**, 45-56.

Flannery, M. S., Snodgrass, R. D. and Whitmore, T. J. (1982). Deepwater sediments and trophic conditions in Florida lakes. *Hydrobiologia*, **92**, 597-602.

Gächter, R., Meyer, J. S. and Mares, A. (1993). The role of microorganisms in the mobilization and fixation of phosphorus in sediments. *Hydrobiologia*, **253**, 103-121.

Gonsiorczyk, T., Casper, P. and Koschel, R. (1995). Vergleichende Sedimentuntersuchungen zum Kohlenstoff- und Phosphorgehalt in baltischen Seen. *Limnologica*, **25**, 365-379.

Gonsiorczyk, T., Casper, P. and Koschel, R. (1997). Variations of phosphorus release from sediments in stratified lakes. *Wat., Air and Soil Polut.*, in press.

Gonsiorczyk, T., Casper, P. and Koschel, R. (in preparation). Phosphorus diagenesis in the bottom sediment of Lake Stechlin.

Hupfer, M. (1995). Bindungsformen und Mobilität des Phosphors in Gewässersedimenten. In: Steinberg, C., Bernhard, H. and Klapper, H. (eds.). *Handbuch „Angewandte Limnologie". Ecomed, Landsberg*, IV-3.2, 1-22.

Hupfer, M. Gächter, R. and Giovanoli, R. (1995). Transformation of phosphorus species in settling seston and during early sediment diagenesis. *Aquatic Sciences*, **57**, 305-324.

Jäger, P. and Röhrs, J. (1990). Phosphorfällung über Calciumcarbonat im eutrophen Wallersee (Salzburger Alpenvorland, Österreich). *Int. Revue ges. Hydrobiol.*, **75**, 153-173.

Kasprzak, P. Benndorf, J. Koschel, R. and Recknagel, F. (1988). Applicability of food-web manipulation in the restoration program of a hypertrophic stratified lake: model studies for Lake Haussee (Feldberg, GDR). *Limnologica*, **19**, 87-95.

Kleeberg, A. and Kozerski, H. P. (1997). Phosphorus release in Lake Großer Müggelsee and its implications for lake restoration. *Hydrobiologia*, **342/343**, 9-26.

Kleiner, J. (1988). Coprecipitation of phosphate with calcite in lake water: A laboratory experiment modeling phosphorus removal with calcite in Lake Constance. *Wat. Res.*, **22**, 1259-1265.

Koschel, R. (1990). Pelagic calcite precipitation and trophic state of hardwater lakes. *Arch. Hydrobiol. Beih. Ergebn. Limnol.*, **33**, 713-722.

Koschel, R., Proft, G. and Raidt, H. (1987). Autochthone Kalkfällung in Hartwasserseen der Mecklenburger Seenplatte. *Limnologica*, **18** (2), 317-338.

Koschel, R., Giering, B., Kasprzak, P., Proft, G. and Raidt, H. (1990). Changes of calcite precipitation and trophic conditions in two stratified hardwater lakes of the Baltic Lake District of the GDR. *Verh. Internat. Verein Limnol.*, **24**, 140-145.

Koschel, R. (1995). Manipulation of whole-lake ecosystems and long-term limnological observations in the Brandenburg-Mecklenburg lake district, Germany. *Int. Revue ges. Hydrobiol.*, **80**, 507-518.

Krienitz, L., Kasprzak, P. and Koschel, R. (1996). Long term study on the influence of eutrophication, restoration and biomanipulation on the structure and development of phytoplankton communities in Feldberger Haussee (Baltic Lake District, Germany). *Hydrobiologia*, **330**, 89-110.

Lopez, P. and Morgui, J. A. (1993). Factors influencing fractional phosphorus composition in sediments of Spanish reservoirs. *Hydrobiologia*, **253**, 73-82.

Mietz, O., Arp, W., Riemer, A., Vietinghoff, H., Henker, K., Wöbecke, K. and Gabrysch, I. (1994). Die Seen im Land Brandenburg. *Bericht des Projektes „Seenkataster Brandenburg". Förderprojekt des Ministeriums für Umwelt, Naturschutz und Raumordnung des Landes Brandenburg.*

Mothes, G. (1980). Die Nährstoffverteilung in Sedimenten norddeutscher Seen und Hinweise auf Austauschprozesse Sediment - Pelagial. *Acta hydrochim. hydrobiol.*, **8**, 35-46.

Murphy, J. and Riley, J. P. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analyt. chim. Acta* **27**, 31-36.

Niessen, F. and Sturm, M. (1987). Die Sedimente des Baldeggersees (Schweiz): Ablagerungsraum und Eutrophierungsgeschichte während der letzten 100 Jahre. *Arch. Hydrobiol.*, **108**, 365-383.

Nürnberg, G. K. (1988). Prediction of phosphorus release rates from total reductant-soluble phosphorus in anoxic lake sediments. *Can. J. Fish. Aquat. Sci.*, **45**, 453-462.

Ostrofsky M. L., Osborne, D. A. and Zebulske, T. J. (1989). Relationship between anaerobic phosphorus release and sedimentary phosphorus species. *Can. J. Fish. Aquat. Sci.*, **46**, 416-419.

Penn, M. R., Auer, T., van-Orman, E. L. and Korienek, J. J. (1995). Phosphorus diagenesis in lake sediments: investigations using fractionation techniques. *Mar. Freshwat. Res.*, **46**, 89-99.

Proft, G. (1984). Die pelagische Calzitfällung und der Carbonatgehalt von Sedimenten pleistozäner Seen. *Acta hydrochim. et hydrobiol.*, **12**: 321-326.

Psenner, R., Pucsko, R. and Sager, M. (1984). Die Fraktionierung organischer und anorganischer Phosphorverbindungen von Sedimenten - Versuch einer Definition ökologisch wichtiger Fraktionen. *Arch. Hyrobiol./Suppl.*, **70**, 111-155.

Psenner, R. and Pucsko, R. (1988). Phosphorus fractionation: advantages and limits of the method for the study of sediment P origins and interactions. *Arch. Hydrobiol. Beih. Ergebn. Limnol.*, **30**, 43-59.

Ramm, K. and Scheps, V. (1996). Phosphorus balance of a polytrophic shallow lake with consideration of phosphorus release. *Hydrobiologia*, **342/343**, 43-53.

Rossknecht, H. (1980). Phosphatelimination durch autochthone Calcitfällung im Bodensee-Obersee. *Arch. f. Hydrobiol.*, **88**, 328-344.

Rybak, J. I. (1969). Bottom sediments of lakes of various trophic types. *Ekol. Pol.*, **17**, 611-662.

Sas, H. (1989). Lake restoration by reducing of nutrient loading. Expectations, experiences, extrapolation. Academic Verlag St, Augustin.

Sass, H., Cypionka, H. and Babenzien, H. D. (1997). Vertical distribution of sulfate-reducing bacteria at the oxic-anoxic interface in sediments of the oligotrophic Lake Stechlin. *FEMS Microb. Ecol.*, **22**, 245-255.

Sinke, A. J. C., Cornelse, A. A., Kelzer, P., van Tongeren, O. F. R. and Cappenberg, T. E. (1990). Mineralization, pore water chemistry and phosphorus release from peaty sediments in the eutrophic Loosdrecht lakes. *Freshwater Biol.*, **23**, 587-599.

Schneider, J., Röhrs, J. and Jäger, P. (1990). Sedimentation and eutrophication history of Austrian Alpine Lakes. In: *Large lakes. - Ecological structure and function*, Tilzer, M. M. and Serruya, C. (Ed.). Brock/Springer Series in contemporary bioscience. Springer-Verlag Berlin, Heidelberg, pp 316-335.

Søndergaard, M., Kristensen, P. and Jeppesen, E. (1993). Eight years of internal phosphorus loading and changes in the sediment phosphorus profile of Lake Søbygaard, Denmark. *Hydrobiologia*, **253**, 345-356.

THE PHOSPHORUS ACCUMULATION AND THE BIOAVAILABILITY OF THE PHOSPHORUS IN LAKE BALATON SEDIMENT

Elemér Dobolyi and Vince Ördög*

Middle-Transdanubian Environmental Inspectorate,
Balatoni u. 6, H-8000 Székesfehérvár, Hungary

*Pannon University of Agricultural Sciences, Institute of Crop Sciences,
Kolbai Károly u. 8, H-9200 Mosonmagyaróvár, Hungary

ABSTRACT

The quality of water in any water body, river, lake, or reservoir is fundamentally affectec by interaction between water and the bottom sediment. This fact is of particular importance in the case of Lake Balaton, Where owing to the shallow depth of 3,25 m on the average, wind generated wave action and other currents result in a much more intensive contact and interaction of the water and sediment phases than in deeper lakes. In order to get a picture on the phosphorus concentration in the sediment in 1978, and again in 1995, undisturbed sediment cores were taken along the longitudinal axis of the lake in eleven points. The comparison of the upper 30 cm layer of the sediment; indicating a much higher (3 times higher) load as we can estimate by measuring point and non-point sources, assuming that the load in the past 17 years was fairly uniform. Until 1978 the 0-5 cm upper layer P-concentration had a value of 618 μg/g; the current measurements are showing a considerable increase with a new average of 788 μg/g. Since neither the total-P, nor the different chemical phosphorus fractions in the sediment reflect the water quality condition above the bottom, an other, a nonchemical parameter was introduced - the so-called biologically available phosphorus (BAP). The determination of BAP is carried out by algal-test in direct contact with the sediment. As test-organism, the cyanobacteria Cylindrospermopsis raciborskii was used. According to the results of the algal-test, it is the sediment BAP which more or less reflects the water quality conditions in the water body above the sediment.

KEYWORDS

bioassay, biologically available phosphorus, Lake Balaton, phosphorus fractionation, sediment phosphorus

INTRODUCTION

In research related to eutrophication in Lake Balaton it is very important to make it clear, whether the bottom sediment acts as a "sink" or "source" of phosphorus compounds, specifically whether under particular conditions the bottom sediment is likely to adsorb phosphorus, or to release it into the aquatic environment. There is no clear, definite answer to this question yet. In this study the results of two phosphorus measurement expeditions will be compared in order to ablich whether or not any phosphorus accumulation occur in the bottom sediment of Lake Balaton. These two expeditions took place in 1978 and in 1995, respectively. Within the framework of this study, another questions should also be answered, namely, whether the quality of sediment in the bottom of the lake is reflecting the quality of the water above it.

METHODS

Sampling, and Methods of Analysis

About 90 % of the top layer of sediment in Lake Balaton consists of extremely soft gray silt of high water content. In order to retrieve the undisturbed sediment core a Beeker type Eijkelkamp sediment sampler was used. The sediment has been examined by 5 cm thick layers down to 30 cm deep. The pore water was removed on the day of sampling by centrifugation and membran filtration..

The analysis of the sediment was made partly from oven dried (105 °C), partly from wet sediment.

Total phosphorus was determined by the method also used in water and wastewater analysis, involving decomposition by sulphuric acid and hydrogene-peroxide; followed by reducing the phosphorus-molybdenum complex with aminonaphtol sulphonic acid. The phosphorus was determined here after by colorimetry (Dobolyi, 1972).

The individual phosphorus fractions were determined during the expedition in 1978 by the method developed by Williams et al (1976); and in the course of the 1995 survey by the method of Hiltjes and Lijklema (1980).

The bioavailability of the sediment was determined by bioassay. The test-apparatus was developed and published by Ördög (1981).

The determination of biologically available phosphorus (BAP) was carried out by bioassay in direct contact with the sediment. For test organism the cyanobacterium, Cylindrospermopsis raciborskii was used. The details of the method were published elsewhere (Ördög and Dobolyi, 1997). In other analytical works the American "Standard Methods" were employed.

SAMPLING POINTS

Samples were taken as a rule along the longitudinal axis of the lake, with points 1, 2, 10 and 11 as exceptions. Sampling point number 1 is situated at the entrance to the Fűzfő bay, about 300 meters from the so-called "steep bank"; while point number 2 is about 200 meters from the sewage treatment plant at Balatonakarattya. (In 1978, during the time of the first sampling the plant discharged waste into the lake; but in the early 1980' s the plant was closed .) Samples 10 and 11 were taken in the Keszthely Bay about 1 km to the north and to the south of the longitudinal lake axis. The sampling points are shown in **Figure 1.**

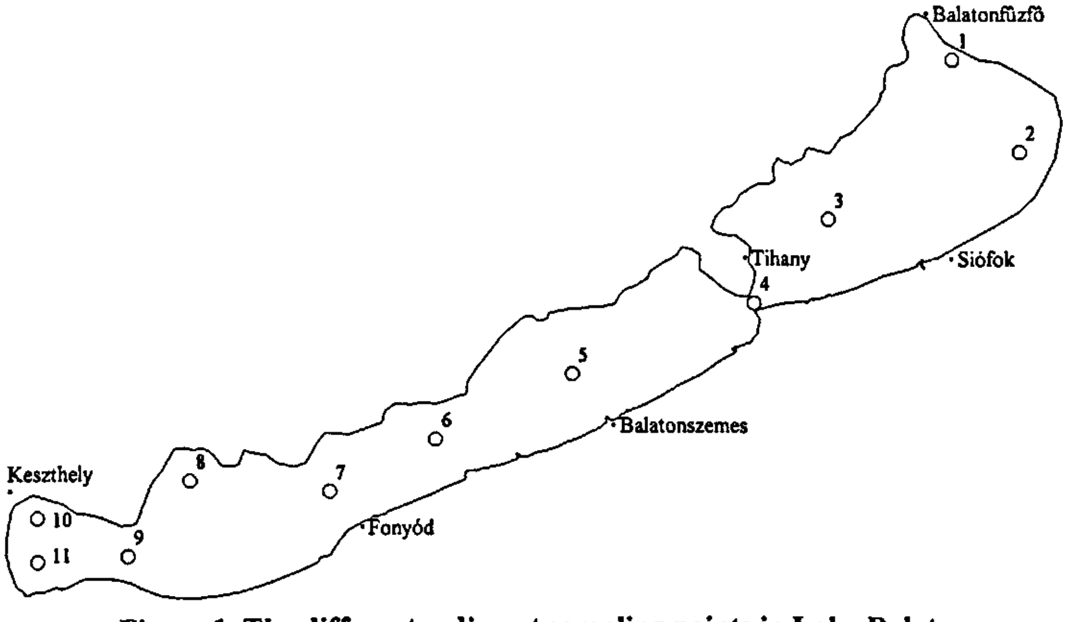


Figure 1. The different sediment sampling points in Lake Balaton

The sampling points were located by GPS; and the same sampling expedition was repeated in 1995. Of the eleven samples, ten consisted of the so-called gray silt; sand was encountered in the Tihany narrows.

RESULT AND DISCUSSION

Detailed analytical results were published elsewhere (Dobolyi, 1980/a). After reviewing the analytical results of the eleven sediment samples in 1978, it was immediately noted that aside from the total P, the Tihany sample differs in all parameters from the similar parameters of all other samples. This is due to the fact that in contrast to the 30-40 per cent SiO_2 content of the other samples the highly sandy sediment at Tihany contains approximately 65 per cent SiO_2. Therefore, this sample has been omitted from the general examination.

The total phosphorus content of the different layers of all eleven sampling points in 1978 is shown in **Figure 2.**

Disregarding the extremely high 945 µg P/g concentration for the Balatonakarattya sample No.2. - presumably a consequence of the sewage treatment plant - the total P in the sediment ranged normally from 500-700 µg P/g.

The total phosphorus data in Lake Balaton sediment in 1995 can be seen on **Figure 3.** The total phosphorus concentration in 1995 was in the range of 600-800 µg P/g dry sediment.

But more expressive is the comparison of the TP concentration in Lake Balaton sediment, if averages of analytical data are calculated on samples taken from identical depth at all sampling points. The P-measurements from 1978 are shown in **Table 1.**, while the results of the 1995 expedition are summerized in **Table 2.**

Sampling depth cm	Dry substance %	CDB extractab-le P /red.Fe-P/ µg/g	NaOH extractab-le P /Al+Fe-P/ µg/g	HCl extractab-le P /Ca-P/ µg/g	Total extractab-le P µg/g	Total P /digested/ µg/g	Total P in unit wet volume µg/ml
0 - 5	26,17	228,6	20,2	226,7	475,5	618,4	161,8
5 -10	31,28	232,1	19,4	216,2	467,7	626,0	195,8
10-15	37,09	194,9	19,0	239,1	453,0	593,0	219,9
15-20	40,67	171,5	19,9	261,1	452,5	580,0	235,9
20-25	41,56	140,8	16,4	281,1	438,3	566,8	235,6
25-30	43,55	138,0	14,2	272,6	424,8	550,7	239,8

Table 1. Averages of analytical data on samples taken from identical depth at all sampling points in 1978.

Sampling depth cm	Dry substance %	NH_4Cl (I+II)ext-ractable P µg/g	NaOH extractab-le P µg/g	HCl extractab-le P µg/g	Total extractab-le P µg/g	Total P /digested/ µg/g	Total P in unit wet volume µg/ml
0-5	25,32	61,8	248	269	557	788	200
5-10	32,25	61,0	187	296	526	758	244
10-15	37,75	63,4	145	288	483	746	282
10-20	41,97	59,9	103	327	495	692	290
20-25	44,32	56,1	108	307	472	725	321
25-30	44,95	54,5	99,5	318	473	675	303

Table 2. Averages of analytical data on samples taken from identical depth at all sampling points in 1995.

As we mentioned earliar, the different phosphorus forms were fractionated by varius methods. The most widely used method of sediment phosphorus fractionation was developed by Williams et al (1976). In this method the following inorganic fractions are distinguished:
Redurctant soluble, Fe-oxide associated-P
Al--and nonreductant Fe-bound-P, and
Ca-bound-P
The first fraction is extracted by a citrate-dithionite-bicarbonate (buffered reducing agent) sulution, mentioned as CDB reagent; the second fraction is extracted by sodiumhydroxide; and the third fraction is by hydrochloride acid. But in the late 1970's practical application of this method was found imperfect in many aspects of the application; and consequently was modified by Hiltjes and Lijklema in 1980. (Hiltjes and Lijklema, 1980.) In the new method still three different fractions are determined; but in the first step, instead of CDB, an ammonium-chloride extraction is applied. It is suggested that in the first NH_4Cl-P fraction, the loosely adsorbed-P can be detected. It is also supposed that it is the NH_4Cl-P fraction which mostly reflects the bioavailable fraction of the total sediment phosphorus. The water quality of Lake Balaton varies in different parts of the lake. The water quality from western Keszthely Bay to eastwards Siofok basin is gradually improving. The reason for this, of course, is the fact that Keszthely Bay receives the highest, while the Siofok basin receives the lowest phosphorus load which are also reflecting in the cholorophyll-a concentrations. During the first sedimetntsampling expedition in 1978, the question - among many other questions - was whether the sediment's quality is reflecting the overlying water body or not. At that time and answer to this question was negative. Meaning, that it is not the sediment, which controls a very low orthophosphate-phosphorus concentration in the lake. The slight changes in the sediment P-concentration show a random pattern over the lake area and depth alike; and do not agree with the major quality differences in the overlying water body. Contrary to **Figure 2**, in **Figure 3** a slightly increasing trend in the sediment-P concentration may be observed in sampling point 1 to 11; but this does not reflect the quality of the overlying water body which changes to a larger degree than the fairly uniform quality of the sediment.

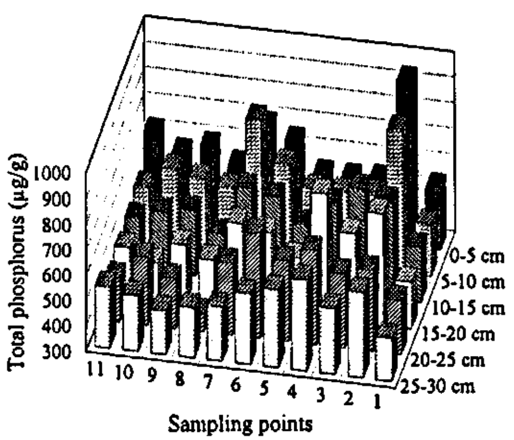


Fig. 2. The total phosphorus concentration in different layers of Lake Balaton sediment in 1978

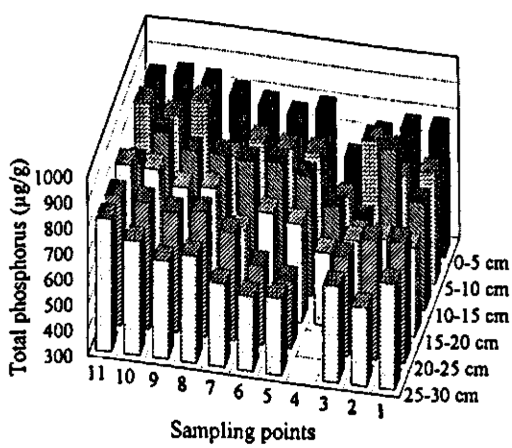


Fig. 3. The total phosphorus concentration in different layers of Lake Balaton sediment in 1995

Similar is the picture of the ammonium-chloride-P fractions of the sediment in **Figure 4.** Nevertheless, it is the NH_4Cl-P fraction which was belived to represent the easily bioavailable fraction of the sediment. Unfortunately, in the case of Lake Balaton sediment, this assumption is not entirely correct. Other NaOH-P and HCl-P sediment fractions were determined and shown in **Figure 5.** and **Figure 6,** but similarly the previous ones, these also provided no useful information.

The pore water of each sediment sample has been analyzed in both expeditions. The results are summarized in **Table 3.**

Sampling depth cm	pH	Conductivity µS/cm	Total dissolved P µg/l	PO_4-P µg/l
		1978		
0-5	8,10	604	191	79,3
5-10	8,17	618	195	96,2
10-15	8,23	606	182	109
15-20	8,25	596	230	131
20-25	8,26	596	199	143
25-30	8,27	533	209	122
		1995		
0-5	7,68	839	535	61,9
5-10	7,86	817	563	97,9
10-15	7,96	800	560	116
15-20	7,97	784	622	154
20-25	8,02	717	643	187
25-30	8,05	682	686	169

Table 3. Averages of analytical data of pore-water taken from identical depth at all sampling points in 1978 and 1995 as well.

Surprisingly, while the ortophosphate-phosphorus concentration (DRP) hardly changed during the years from 1978 until 1995, the concentration of totally dissolved phosphorus has increased approximately three times. This fact seems to justify the published statement by Dobolyi in 1980, that the alga-produced condensed phosphate was not directly available for the next generation of alga; therefore, the enrichment in the pore water can be expected. Namely, the gap between the totally dissolved-P and the ortophosphate-P in pore water is the sum of organic and condensed phosphates (Dobolyi, 1980/b).
Since neither the total-P nor the different phosphorus fractions in the sediment reflect the water quality condition in the water body above the sediment, an other non-chemical parameter was introduced - the biologically available phosphorus or BAP.
Regarding the eleven sampling points, only the first (0-5 cm), the third (10-15 cm), and the fifth (20-25 cm) layers were tested. See **Figure 7.**

The bioavailability of the sediment in the upper 0-5 cm layer varies between 96,5 µgP/g and 288,4 µgP/g. These values much higher than the NH_4Cl-P fraction which ranges from 33,4 µgP/g to 101 µgP/g. According to the results of the biosaasy, it is the sediment BAP which more or less reflects the water quality condition in the water body above the sediment.

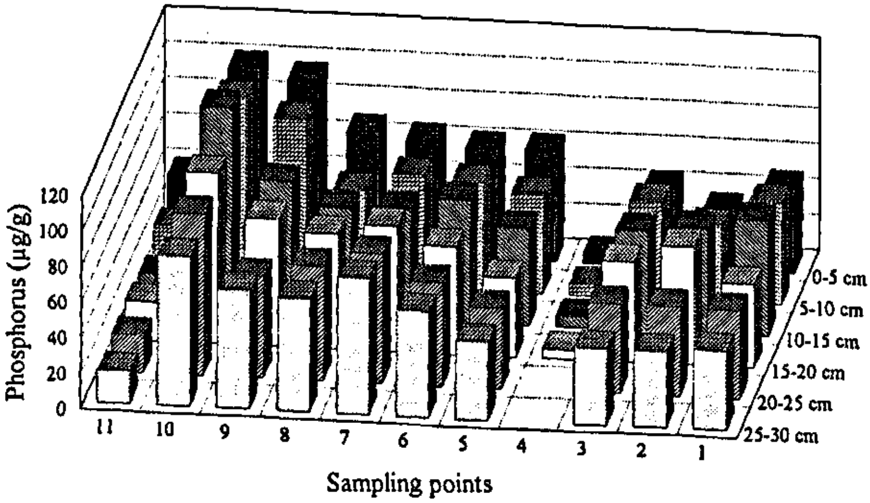


Figure 4. The extractable fraction of phosphorus by ammonium chloride in different layers of Lake Balaton sediment in 1995

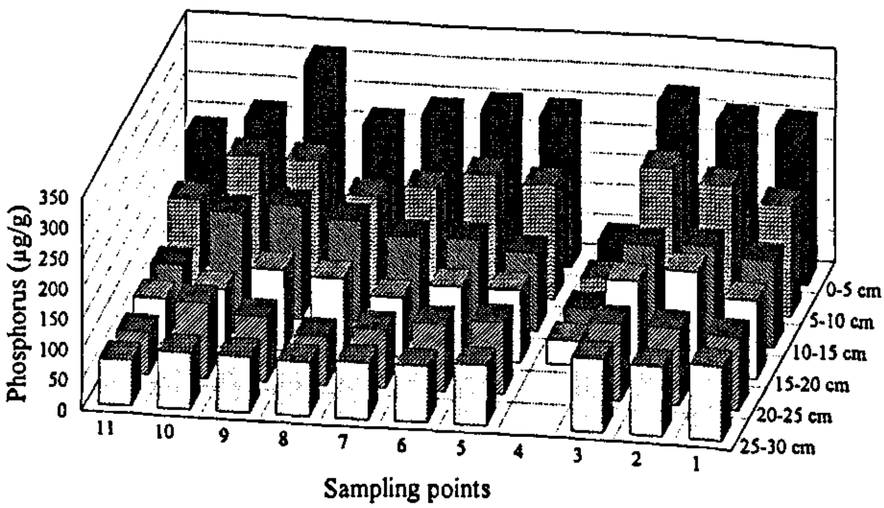


Figure 5. The extractable fraction of phosphorus by sodium hydroxide in different layers of Lake Balaton sediment in 1995

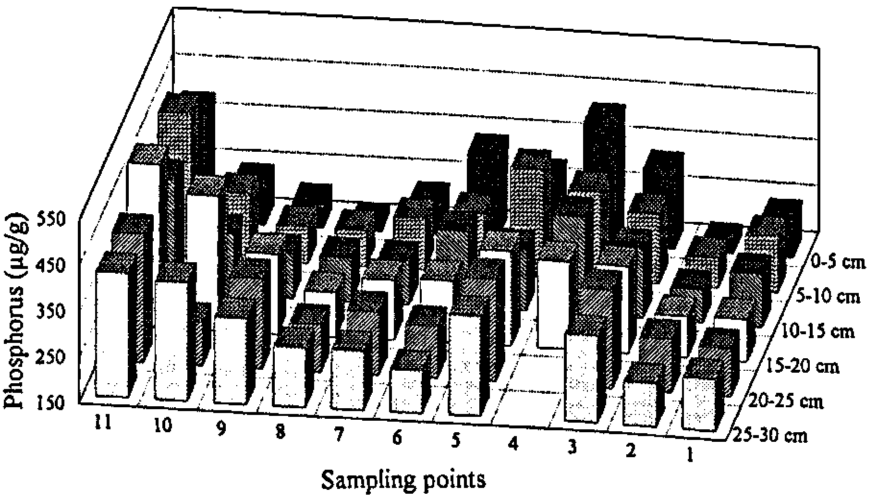


Figure 6. The extractable fraction of phosphorus by hydrogen chloride in different layers of Lake Balaton sediment in 1995

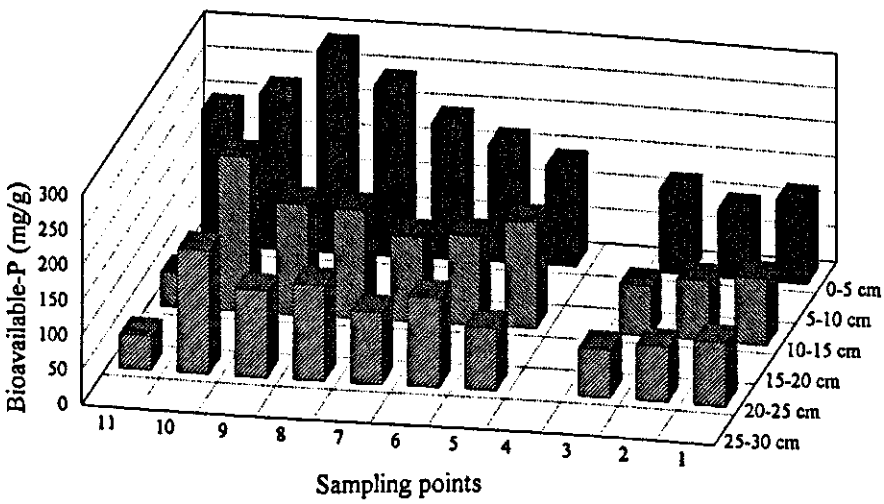


Figure 7. The biologically available phosphorus concentration in different layers of Lake Balaton sediment in 1995

CONCLUSIONS

In order to be able to control the eutrophication process of the lake, the first and most important step is to determine the phosphorus load. In the case of Lake Balaton the responsible authorities and research institutions have never measured an external phosphorus load greater than 150-200 t/a. But it was always suspected that this P-load value is very much underestimated. At the conclusion of this sediment survey this assumption became even more evident. Between the two sediment expeditions in 1978 and in 1995, the phosphorus accumulation in the Lake sediment - according to **Table 1** and **Table 2** - was about 10.000 ton. If we assume that during the 17 years between 1978 and 1995 the load was uniform, it would indicate an approximate 580-590 t annual load. This load is about three times higher than we have ever measured. This fact is underlines the paramount importance of a more accurate measurement of the external phosphorus load of the lake.

With regards to the question whether or not the bottom sediment is a sink or surce of the phosphorus - on the whole - the answer is obvious since there is a steady P accumulation during the last decades. Whether the sediment temporarily can act as a source of phosphorus it is also quite obvious and well known, even though the extent of this internal load is still unknown.

Since neither the total P nor the different chemical phosphorus fractions in the sediment reflect the water quality condition above the botton, the biologically available phosphorus parameter was introduced. This has shown a more or less conformity with the water quality conditions in the water body above the sediment. Consequently, it seems that the bioassay is the only parameter with which the bioavailability can be estimated.

REFERENCES

Dobolyi, E. (1972) Total phosphorus determination in wastewaters (Hungarian) *Hidrológiai Közlöny*, **52**, 81-86.

Dobolyi, E. (1980/a) Data on the bottom sediment in Lake Balaton *Proceedings of the Second Joint MTA/IIASA Task Force II. Meeting on Lake Balaton Modeling*, **II**. Aug. 27-30. 1979.

Dobolyi, E. (1980/b) Identification of different phosphorus forms and their role in the eutrophication process of Lake Balaton, *Developments of Hydrobiology*, **vol. 2**, (ed by J. Barica and L. R. Mur), 13-22.

Hiltjes, A.H.M. and L.Lijklema, L.(1980) Fractionation of inorganic phosphates in calcareous sediments. *J.Environ.Qual.*, **92**, 491-500

Ördög, V.(1981) Apparatus for laboratory algal biossay. *Int.Revue ges. Hydrobiol.*, **67**, 127-136.

Ördög, V. and Dobolyi, E.(1997) The efficiency and justifiability of thin-layer dredging in the Keszthely bay (Lake Balaton) in order to remove phosphorus. *Proceedings of this Conference*

Williams, J.D.H., Jaquet, J.M. and Thomas, R.L.(1976) Forms of phosphorus in the surficial sediments of Lake Erie. *J.Fish, Res. Board Can.* **33**, 413-429.

PHOSPHORUS UPTAKE AND ACTIVE GROWTH OF *ELODEA CANADENSIS* MICHX. AND *ELODEA NUTTALLII* (PLANCH.) ST. JOHN

Anna H. Eugelink

Dept. of Water Quality Management and Aquatic Ecology, Agricultural University Wageningen, P.O. Box 1029, 6870 DA Renkum, The Netherlands.

ABSTRACT

Two submerged freshwater macrophytes (*Elodea nuttallii* (Planch.) St. John and *Elodea canadensis* Michx.) were used in these experiments. The option to perform both indoor and outdoor experiments with active growth of the two *Elodea* species, was very useful to compare growth rates and routes of phosphorus uptake, translocation and possible excretion. The growth experiments show that *Elodea nuttalli* has a higher growth rate than *Elodea canadensis* both in the field and under laboratory conditions.
The uptake and translocation of phosphorus was studied using ^{32}P in a partitioned container. Roots and leaves were supplied with ^{32}P separately and simultaneously. Both macrophytes were able to take up phosphorus both with leaves and roots. The uptake rate of ^{32}P by roots was higher in *Elodea nuttallii* when tracer was supplied to the root compartment only. Leaf uptake was stimulated by the supply of phosphate to both compartments, the uptake was faster and reached a higher level than when the tracer was injected to the leaf compartment only. A proportion of the ^{32}P uptake by either roots or leaves was translocated. Shoot-to-root translocation predominated over the reverse. There was no significant differences between both species in root uptake rates, but leaf uptake was significantly higher in *Elodea canadensis*. The two studied *Elodea* species do not have a strong preferential source for phosphorus. They act as a sink collecting their phosphorus from water or sediment.

KEYWORDS

growth-rate; radio-active tracer; submerged macrophytes; translocation; uptake-rate.

INTRODUCTION

Changes in nutrient loading to lakes are one of the most important reasons for fluctuations in areal cover of submerged macrophytes. At increasing nutrient concentrations, submerged macrophytes decline due to shading by phytoplankton and epiphytes (Phillips et al., 1978).

Elodea canadensis Michx. was first found in 1859 in The Netherlands (van der Meijden, 1990). The plant is found in eutrophic calcareous water (pH 6.5-10) and cool conditions, optimum water temperature is 10-25 °C. It often builds large single species stands and may be dominant in waters 0.1-1.5 m deep (Cook and Urmi-König, 1985). At the onset of cold weather *Elodea canadensis* dies back; it often turns black in autumn. Spring regrowth is from underground stems crowned by roots or winter buds (turions). There is no dormancy. Growth recommences as soon as the temperature rises. The plant is very brittle and fragments easily.

Elodea nuttallii (Planch.) St. John is also native in North America. The species is largely similar to *Elodea canadensis*, but seems to be concentrating somewhat more southward. In Europe *Elodea nuttallii* was first collected in 1939 (Belgium). In 1941 the macrophyte was found in The Netherlands. Female *Elodea nuttallii* is actively spreading in many parts of Europe and seems to be replacing *Elodea canadensis* in many localities (Cook and Urmi-König, 1985). *Elodea nuttallii* grows in calcareous waters in lakes, ponds, slowly flowing streams and canals. From December or January to March it overwinters as prostrate shoots with green leaves that form dense mats on the bottom; these shoots grow when the temperature exceeds 4°C. From September onwards the branched apical shoots become detached and sink in November and December. The life cycle ends with dormancy on the bottom in the winter.

A central question in the nutritional ecology of submerged aquatic macrophytes has been the source of their nutrients. Macrophytes have access to two different nutrient sources, e.g. sediment and water column.

Frank and Hodgson (1964) were among the first to devise a two chambered system to actually test the uptake and translocation of nutrients in submerged macrophytes. Radio-isotope labelled tracer was supplied to one of the chambers. Spence (1972) suggested that two factors are controlling the occurrence of submerged macrophytes. In nutrient-rich waters the relationship between algae and higher plants is controlled by the turbidity of the water, whereas in nutrient-poor waters the potential for uptake of minerals through the roots or analogous organs (e.g. rhizomes) determines the presence of macrophytes. Bole and Allan (1978) and Carignan and Kalff (1980) found that, at least for *Myriophyllum spicatum*, the foliar uptake of phosphorus becomes more important as phosphorus concentrations in the water increase. So there is evidence that these species act as opportunistic species by taking phosphorus from the best available source.

The aim of this study was to provide information on differences in growth-rate between *Elodea canadensis* and *Elodea nuttallii*. The option to perform both indoor and outdoor experiments with active growth of the two *Elodea* species, was very usefull to compare growth rates and routes of phosphate uptake, translocation and possible excretion. The field experiments were performed in artificial ditches (Portielje, 1994) in which the response of the vegetation towards different nutrient loading levels was studied during 7 years. To understand the development of the vegetation in the ditches it was necessary to know the pathway of phosphorus and which organ is most efficient at nutrient uptake. Understanding their behaviour with respect to nutrients ultimately may help in the development of control strategies.

MATERIAL AND METHODS

Plant material (growing tips of both species: 5 cm long) and water needed for the experiments were collected from outdoor artificial ditches (Eugelink, submitted). The water was filtered through Whatman GF/C filters prior to use in order to remove macroalgae. For the indoor growing experiments thirty-five apical shoots were selected, each shoot was marked at the beginning of the experiment by a small numbered thread approximately 2 centimeters from the apex. At the same time *Elodea* shoots were marked in the ditches for the outdoor in situ experiments. The experiments were done early July 1995. Both indoor and outdoor experiments were performed at a temperature ranging from 20.2 - 23.4 °C and an average light intensity at the surface of 425 $mE.m^{-2}.s^{-1}$ with 16 hours photoperiod (indoor: Philips 400 W HPIT metal halide lamp). Stem length of five shoots from each species were recorded after 1, 5, 10, 15, 20, 25 and 30 days. The original main shoot length was measured, lateral shoot and root numbers growth were counted.

For the phosphorus uptake experiments partitioned containers were used (Figure 1). The plants were potted with the stems leaving the root compartment through a tightly fitted stopper filled with lanolin to form a water-tight seal around the stem without constricting it. Separation of shoots and roots made it possible to expose either plant part to radioactive P-tracer introduced into individual compartments. The submerged leaves received light whereas the underground parts were placed in the dark. Only fresh green plants without visible epiphytes (whole plants were washed in 0.01 N $KMnO_4$ to remove epiphyton) and of approximately the same size (± 10 cm) were used. Experiments were carried out in a climate room with a day/night rhythm of respectively 14 hours light at 21°C and 10 hours dark at 15°C. The light-conditions were 0 at night and 160 $mEm^{-2}s^{-1}$ at day. Initially, 10 experimental chambers were prepared with filtered ditch water. The water was added to the compartments with a total concentration of 300 mg PO_4-$P.l^{-1}$ at a pH of 8.0 ± 0.4. Carrier-free ^{32}P was injected by means of a syringe into one of the selected compartments. Insertion of the radio-active tracer was at the beginning of the light period for all experiments. The plants were harvested after 1, 2, 4, 12, 24, 48 and 72 hours of incubation. Plant material recovered from the compartments was washed with distilled water, to remove adsorbed radio-isotopes, and blotted with paper towels. The roots were separated from the stem and the leaves. The upper plant parts were further severed into stems and leaves. The activity of all samples was counted using liquid scintillation techniques (Beckman LS 6000 TA). The activity was corrected for background and isotope decay. Plant material was counted in fractions in vials with deionized water. The uptake and the translocation was studied after root- and leaf-chambers were spiked simultaneously as well as separately. The initial activity of the ^{32}P in the solution was 10 mCi.

RESULTS

Growth-rate

The difference in growth between indoor shoots is shown in figure 2A. *Elodea canadensis* showed almost no stem elongation after 15 days and had grown to only about 6.5 cm after 30 days. Conversely, *Elodea nuttallii* showed a remarkable elongation of the stem. Growth started almost immediately, fastest growth was observed between 10 and 15 days, followed by further, although smaller increase. The difference in stem length between the two *Elodea* species was highly significant ($P<0.001$) on each sampling date.

The growth results of the outdoor experiments are shown in figure 2B. *Elodea canadensis* in this outdoor experiment showed a greater stem elongation than in the indoor experiments. Fastest growth was exhibited between 1 and 15 days, followed by lower increase rate. *Elodea nuttallii* showed a smaller growth rate outdoor than in the indoor experiments but it was still higher than for *Elodea canadensis*. The number of auxillary stems was higher in the outdoor experiment for both species. The number of branches was higher for *Elodea nuttallii*.

Regular observations (personal) were also made of the time taken for new roots to appear on

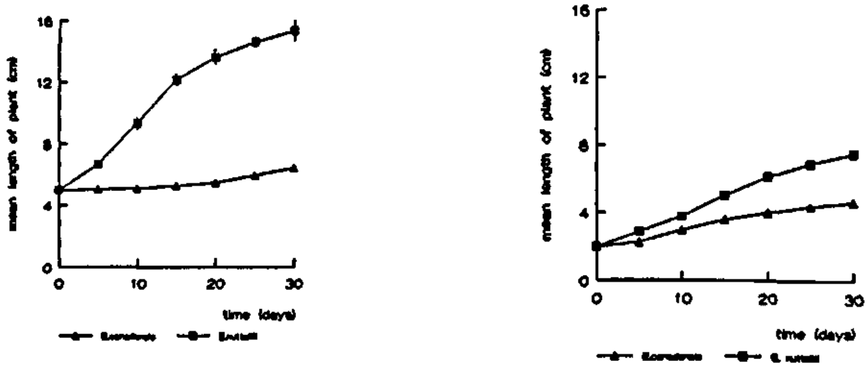


Fig. 2. Growth of Elodea canadensis and Elodea nuttallii under indoor and outdoor conditions

the stem pieces. This was shorter in *Elodea nuttallii*. The adventitious roots emerged from the same nodes where the lateral shoots had developed and their formation was mostly accompanied by the lateral shoot development. Under indoor conditions, most of the initial roots had developed in one week after the cutting of the mother shoot, and secondary roots successively began to emerge from about two weeks after the cutting. In contrast, the roots under outdoor conditions developed slowly and no secondary roots emerged.

Uptake of phosphorus in Elodea nuttallii

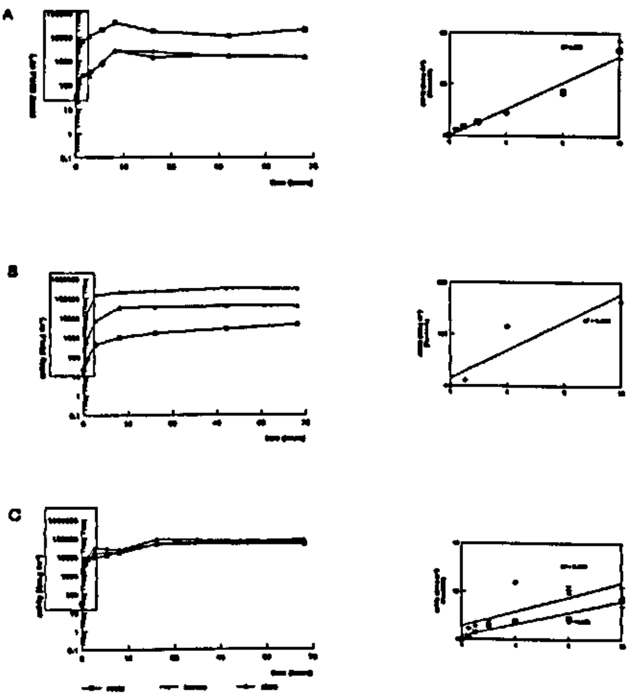
The uptake of ^{32}P in roots of *Elodea nuttallii* and translocation to leaves during incubation time is shown in figure 3A. Uptake started with a low uptake rate but became more rapid after six hours of incubation. The correlation coefficient for an assumed lineair rate of uptake for the first 12 hours was R^2=0.978. The uptake of phosphorus by leaves is shown in figure 3B. Phosphate uptake by leaves was more rapid than the uptake by roots in the first four hours. The correlation coefficient for linear regression on the initial uptake was R^2=0.809. After four hours leaf uptake the translocation to the roots decreased coinciding with a dark period. During the second light period a slight increase occurred. The proportion of phosphorus translocated to the roots (Figure 3B) was significantly greater ($p < 0.05$) than the upward translocation after root uptake (Figure 3A). Uptake of phosphorus by both leaves and roots was observed when tracer was injected into both compartments (Figure 3C). Shoot uptake was slightly greater but not significant. After four hours leaf uptake decreased during by a dark period, but root uptake continued to increase. This figure (3C) shows that translocation is due to leaf uptake because stem activity followed the decrease of leaf uptake. After this dark period an increase was shown for the next light period.

Release of translocated phosphorus from shoots or roots to the surrounding water could not be detected in any of the experiments: no increase of ^{32}P activity in the water was measured.

Uptake of phosphorus in Elodea canadensis

The uptake of phosphorus by roots of *Elodea canadensis* and translocation to leaves during

Figure 3:



incubation time is shown in figure 4A. The correlation coefficient for the regression of uptake rate was R^2=0.986 for the first 12 hours. Translocation to the leaves by *Elodea canadensis* was significantly greater than in *Elodea nuttallii* (Figure 4A and 3A). The uptake by leaves is shown in figure 4B (R^2=0.922). The uptake rate was significantly greater than in *Elodea nuttallii*. Translocation to the roots was slower than in *Elodea nuttallii* (Figure 3B). When tracer was injected into both (Figure 4C) compartments leaves showed a greater phosphorus uptake than roots, but not significant (R^2=resp. 0.881 and 0.994).

Release of translocated phosphorus from shoots or roots to the surrounding water could not be detected.

DISCUSSION AND CONCLUSIONS

Internal eutrophication processes play an important role in the eutrophication of surface waters and as well for the growth and distribution of aquatic macrophytes. Some aquatic macrophytes, such as *Elodea* species, largely reproduce by means of vegetative reproduction.

Canopy formation is of particular importance to submerged macrophytes. Such a feature is characteristic for *Elodea* and related genera. Canopy production bestows an obvious advantage as it severely restricts the competitive ability of other species (Titus and Adams, 1979; Barko and Smart, 1981b).

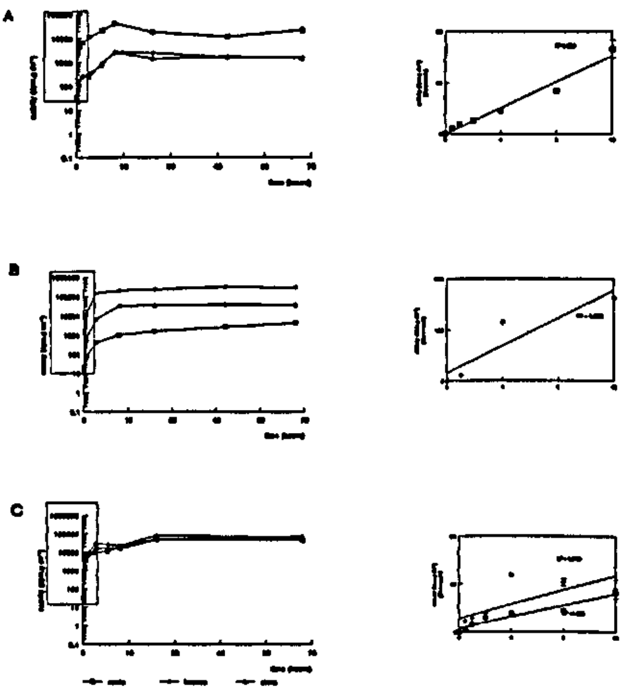


Fig. 4. Phosphorus uptake by Elodea canadensis (mean values, n=3).

Present growth experiments show that *Elodea nuttallii* has a higher growth rate (than *Elodea canadensis*) both in the field and under laboratory conditions. Both species can form a canopy. If stem elongation and auxillary stem production is more rapid in *Elodea nuttallii*, this species may produce a canopy more quickly than *Elodea canadensis*. Results from earlier observations in the outdoor artificial ditches indeed showed that *Elodea nuttallii* started growth early in the growing season (Eugelink, submitted).

The present investigation showed that both *Elodea* species are able to take up ^{32}P through both roots and leaves (Figure 3A/B, 4A/B). A proportion of the phosphorus uptake by either roots or leaves was translocated.

It has been suggested that the nutrient uptake by rooted aquatic macrophytes occurs primarily from the substrate with the highest nutrient concentration (Denny, 1972).
In our experiments the uptake rate of ^{32}P by roots was higher in *Elodea nuttallii* when tracer was supplied to the root compartment only (Figure 3A and 3C). Leaf uptake was stimulated by the supply of phosphate to both compartments, the uptake was faster and reached a higher level than when the tracer was injected to the leaf compartment only (Figure 3B/C). Shoot-to-root translocation predominated over the reverse.

Uptake of ^{32}P by roots was as high and fast when supplied to either the root compartment or to both compartments for *Elodea canadensis* (Figure 4A/C). Leaf uptake was higher when only the leaf compartment was spiked (Figure 4B/C). ^{32}P uptake by leaves was significant higher than by roots, when supplied to either the root compartment or to the shoot compartment. Root-to-shoot and shoot-to-root translocation are of the same magnitude in *Elodea canadensis*. There was no significant difference between both species according to root uptake (Figure 3A and 4A), but leaf uptake was significant higher by *Elodea canadensis* (Figure 3B and 4B). All leaf uptake experiments show a decrease in uptake rate in a dark period and an (slight) increase in the next light period.

DeMarte and Hartman (1974) studied *Myriophyllum exalbescens* and found uptake by both leaves and roots and translocation in both directions as well (root-shoot, shoot-root). Quantitative estimates of phosphorus movements were not provided. They concluded that in *Myriophyllum spicatum* L. and *Egeria densa* Planch. phosphate can be absorbed by different organs, but the absorption by roots is much faster than by leaves, and the lowest for stems. From Denny's (1972) work, the conclusion may be drawn that both the sediments and the water can supply nutrients. The relative quantities absorbed from these two sources depend on the plant species. The possession of an active transport system as in *Elodea* caused by root pressure may be an advantage for those species capable to utilize both leaves and roots over species, lacking this mechanisms, e.g. *Ceratophyllum*. Carignan and Kalff (1980) showed that nine species of freshwater aquatic macrophytes depend almost exclusively on the sediments for their phosphorus supply. In addition, many studies have indicated a net leakage of phosphorus from the leaves to the surrounding water (McRoy et al., 1972; DeMarte and Hartman, 1974; Twilley et al., 1977).

Differences between the results of the present study and those cited above can be due to actual differences between species of rooted aquatic macrophytes. Welsh and Denny (1979) found translocation of phosphorus in both directions in two species of *Potamogeton* but negligible excretion. Bole and Allan (1978) similarly found negligible leakage from shoots to water by *Myriophyllum* and *Hydrilla*. Barko and Smart (1980, 1981) studied six submergent macrophytes and found little or no phosphorus excretion by actively growing plants.
The literature thus provides conflicting evidence as to leaf excretion of phosphorus by actively growing rooted macrophytes. In this study actively growing *Elodea nuttallii* and *Elodea canadensis* did not excrete phosphorus into the water. This means that both *Elodea* species do not act as a 'nutrient pump' under the prevailing experimental conditions by taking up phosphorus from the sediment by roots, transporting it to the shoots and excrete the phosphorus into the surrounding water.
As with other submerged aquatic macrophytes that have been studied, both *Elodea* species were capable of taking up nutrients through both leaves and roots. Leaf uptake was affected by the supply to the roots in case of *Elodea nuttallii*.

The two studied *Elodea* species do not have a strong preferential source for phosphorus. They act as a sink collecting their phosphorus from water or sediment. The highest phosphorus content was found in *Elodea nuttallii* (Eugelink, submitted). According to the growth experiments *Elodea nuttallii* length growth is greater (significantly under laboratory conditions) than length grwoth of *Elodea canadensis*. Faster and higher phosphorus uptake rates results in lesser length growth for *Elodea canadensis*.

REFERENCES

Barko, J.W., and Smart, R.M. 1980. Mobilization of sediment phosphorus by submerged freshwater macrophytes. Freshwater Biol. 10: 229-238.

Barko, J.W., and Smart, R.M. 1981a. Sediment based nutrition of submerged macrophytes. Aquat. Bot. 10: 339-352.

Barko, J.W. and Smart, R.M., 1981b. Comparative influences of light and temperature on the growth of selected submersed freshwater macrophytes. Ecol. Monogr. 51: 219-235.

Bole, J.B., and Allan, J.R. 1978. Uptake of phosphorus from sediment by aquatic plants, *Myriophyllum spicatum* and *Hydrilla verticillata.* Water Res. 12: 353-358.

Carignan, R., and Kalff, J. 1980. Phosphorus source for aquatic weeds: water or sediments? Science (Wash.DC) 207: 987-989.

Cook, C.D.K. and Urmi-König, K., 1985. A revision of the genus *Elodea* (Hydrocharitaceae). Aquatic Botany 21: 111-156.

DeMarte, J.A., and Hartman, R.T. 1974. Studies on absorption of ^{32}P, ^{59}Fe, and ^{45}Ca by water-milfoil (*Myriophyllum exalbescens* Fernald). Ecology 55: 188-194.

Denny, P. 1972. Sites of nutrient absorption in aquatic macrophytes. J. Ecol. 60: 819-829.

Eugelink, A.H. Response of vegetation to nutrient loading in shallow aquatic ecosystems (submitted).

Frank, P.A. and Hodgson, H., 1964. A technique for studying absorption and translocation in submersed plants. Weeds 12, 546-555.

McRoy, C.P, Barsdate, R.J. and Nebert, M., 1972. Phosphorus cycling in an eelgras (*Zostera marina* L.) ecosystem. Limnol. Oceangr. vol. 17(1), 58-67.

Meijden, van der R., 1990. Heukels' Flora van Nederland. Wolters-Noordhoff, Groningen, 21ste druk, 662 p.

Nichols, S.A. and Shaw, B.H., 1986. Ecological life histories of the three aquatic nuisance plants, *Myriophyllum spicatum, Potamogeton crispus* and *Elodea canadensis.* Hydrobiologia 131: 3-21.

Spence, D.H.N. 1972. Light on freshwater macrophytes. Trans. Bot. Soc. Edinburgh 41: 491-505.

Titus, J.E. and Adams, M.S., 1979. Coexistence and comparative light relations of the submersed macrophytes *Myriophyllum spicatum* L. and *Vallisneria americana* Michx. Oecologia 40: 273-286.

Twilley, R.R., Brinson, M.M. and Davis, G.J. 1977. Phosphorus absorption, translocation, and secretion in *Nuphar luteum.* Limnol. Oceanogr. 22: 1022-1032.

Welsh, R.P.H. and Denny, P. 1979. The translocation of phosphorus-32 in two submerged aquatic angiosperm species. New Phytol. 82: 645-656.

BINDING CONSTANTS OF CHLOROBENZENES AND POLYCHLOROBIPHENYLS FOR ALGAL EXUDATES

Albert A. Koelmans and Evelyn H.W Heugens

Department of Water Quality Management and Aquatic Ecology, Wageningen Agricultural University, P.O. Box 8080, 6700 DD Wageningen, The Netherlands

ABSTRACT

In eutrophic ecosystems, primary production may significantly affect the behaviour of toxicants or the accumulation in the aquatic food chain. Little is known about the potential of extracellular carbon, eg., exudates, to scavenge hydrophobic chemicals. In a laboratory *Selenastrum capricornutum* culture with 30% extracellular carbon, phytoplankton-water bioconcentration factors (*BCF*) and exudate-water association factors (K_{DOC}) were measured for two chlorobenzenes and 4 PCBs (total $\log K_{ow}$ range 4.6 - 7.4). The apparent log*BCF*s were measured after 3 day batch equilibration and varied linearly with $\log K_{ow}$ with a slope of 0.63. The K_{DOC} values for exudates from the same batch were measured using a gas purge method and were very high and more or less equal to K_{ow}: $\log K_{DOC}=\log K_{ow}$ - 0.057 (r2=0.926, n=6). The strong association with exudates limits the bioavailability of PCBs in laboratory cultures and natural aquatic systems.

KEY WORDS

Algae; bioconcentration; bioavailability; eutrophication; exudate; polychlorobiphenyls; sorption

INTRODUCTION

In eutrophic systems the fate and food chain accumulation of hydrophobic chemicals (HOCs) are highly influenced by bioconcentration in phytoplankton. Algal blooms provide a considerable mass of organic carbon in the water column with a high HOC scavenging capacity. Algae and detritus are the primary entrance for HOCs in the food chain. Dependent on the growth conditions (trophic state, temperature), a considerable percentage of carbon fixation may be as extracellular carbon. Extracellular release amounting to 95% of the total carbon fixation has been reported (Fogg, 1971). Besides planktonic particles, the extracellular carbon represents a highly dynamic carbon pool, possibly with a large binding capacity for HOCs. As such, exudation may affect the cycling and bioavailability of HOCs and may represent an additional HOC loss mechanism from aquatic organisms. Such mechanisms are not incorporated in the current fate or food chain models.

Besides a reduced bioavailability in natural waters, DOC-HOC associations play a key role in the current debate on the interpretation of laboratory bioconcentration data for phytoplankton. A significant binding to extracellular carbon may explain the observed maximum in log*BCF*-$\log K_{ow}$ plots above K_{ow}=6.5 (Swackhamer and Skoglund, 1993; Stange and Swackhamer, 1994; Koelmans *et al*, submitted), the observed dependency of *BCF* on DOC concentrations in the medium (eg., Richer and Peters, 1993), or the dependency of *BCF* on algal density (Sijm *et al*, 1995). However, none of these authors provide evidence for this phenomenon. To date, no data are available on the HOC binding capacity of extracellular carbon exudated by freshwater algae.

In this paper direct evidence for binding of PCBs to Selenastrum capricornutum exudates is reported. *BCF*s in a laboratory culture containing 30% algal exudates were measured by traditional batchwise equilibration. The exudate containing medium was separated and placed in a gas sparging apparatus to strip the associated PCBs and measure association constants. The association constants were used to correct the apparent *BCF*s measured by batch equilibration.

MATERIALS AND METHODS

Chemicals - 1,2,3,4-Tetrachlorobenzene (TeCB, >96%) was obtained from Merck; 1,2,4,5-Tetrachlorobenzene (98%) was obtained from Aldrich; hexachlorobenzene (HCB, >98%) was obtained from BDH Chemicals, and 2,4,4'-trichlorobiphenyl (PCB28, 99.9%), 2,2',5,5'-tetrachlorobiphenyl (PCB52, 99.2%), 2,3',4,4',5'-pentachlorobiphenyl (PCB118, 99.7%), and 2,2',3,4,4',5,5'-heptachlorobiphenyl (PCB180, 99.8%) were obtained from Promochem. Nanograde or *p.a.* quality organic solvents were obtained from Promochem (acetone, diethyl ether) and Mallinckrodt (2,2,4-trimethylpentane).

Algal culturing - A culture of *Selenastrum capricornutum* (NIVA-CHL1, *Rhaphidocelis subcapitata*) was kindly supplied by the Norwegian Institute for Water Research (NIVA). The cultures were scaled up under continuous light ($100 \mu E \times cm^{-2} \times s$, surface) in COMBO mineral medium (Kilham *et al*, 1995), prepared axenicly in 0.2μm filtrated Nanopure water (Sybron-Barnstead, Dubuque, IA). The light regime was 16:8 L:D at $100 \mu E \times cm^{-2} \times s$ (surface). Temperature was 20±2°C. Bioconcentration experiments were performed after the stationary growth stage has been reached.

Monitoring of algal density during culturing and bioconcentration - Dry weight (DW, mg/L) was measured by filtration over 0.45 μm (glass fibre, Whattmann, GF/C) and drying the residues at 105°C till constant weight. Cell numbers and particle size distribution (PSD) were measured with a Coulter Multisizer II (Coulter Electronics, Mijdrecht, The Netherlands). Total organic carbon was measured in the algal suspensions (TOC, mg/L) and in the centrifugated (5 min., 2000 g) medium (DOC, mg/L) with an OIC (College Station, TX) Model 700 TOC analyzer. Organic carbon fractions of the algal cells were calculated as (TOC-DOC)/DW. After phase separation the absence of algal cells in the supernatants was ascertained by microscopic inspection (Nikon, type 102, 400×).

Bioconcentration in algae - A five litre culture was spiked with 50 μL of a solution of TeCB, HCB, PCB28, PCB52, PCB118, and PCB180 in acetone. The spiked suspensions were incubated for three days at 20±2°C). After incubation subsamples were taken for triplicate determination of cell numbers, dry weights, and TOC concentrations. The algae were separated from the medium by filtration (0.45μm, glass fibre filters). After phase separation, triplicate HOC concentrations were determined in the algae, and in the medium. A subsample of the medium containing extracellular carbon was obtained by centrifugation (5 min, 2000 g) and used for determination of TOC and cell numbers, and for the subsequent gas purge experiments.

Association with exudates - The gas purge procedure was described before (Koelmans *et al*, 1995), and is repeated here briefly. The 0.8 L subsample from the equilibrated culture was placed in a glass stoppered bottle. The HOCs were purged onto 12 Tenax traps (40 - 60 mesh, Chrompack, The Netherlands) in two weeks, at a flow of 0.44 L/min under continuous stirring at T=20±0.1°C. The gas flow (constant within 1.5%) was adjusted with flow controllers and was passed through vessels containing water, to prevent volume reduction in the algal suspensions. Tenax traps (trap efficiency

100%) were replaced at incremental intervals, and consisted of glass tubes containing glass wool and approximately 0.2 g Tenax. After purging, HOCs, cell numbers and DOC in the medium were analyzed to be able to account for remaining HOC fractions, and DOC losses.

HOC cleanup and analysis - Water samples were extracted with hexane (5:1). The filters with algae were cut into bits and heated under reflux for 90 min. with 2:1 water:hexane. HOCs were eluted from the Tenax columns with diethyl-ether. The extracts were concentrated under a stream of nitrogen (water extracts, Tenax extracts) or with Kuderna-Danish (algae extracts), and cleaned with deactivated alumina and silica. All concentrations were corrected for less than 100% clean-up recoveries of the test compounds, which were determined in triplicate. All samples contained 1,2,4,5-Tetrachlorobenzene as an internal standard to reduce the error of subsequent GC-analysis. A Hewlett-Packard (HP; Avondale, PA) 5890 double-column gas chromatography system equipped with two ^{63}Ni electron capture detectors (ECD) was used in chlorobenzene and PCB quantisation.

Data analysis - For a full discussion of the models used we refer to earlier reports (Koelmans *et al*, 1993; Koelmans *et al*, 1995) . The gas purge induced elimination of HOCs from the aqueous phase is described as an irreversible first-order volatilization process (Hasset and Millicic, 1985):

$$dC_W/dt = -k_v \times C_W \quad \wedge \quad k_v = \frac{F \times H}{RT \times V_w} \tag{1}$$

with t is time (day), C_W (μg/L) is the solution-phase HOC concentration referenced to the water volume V_w (L), k_v is a first-order volatilization rate constant (day^{-1}), F is the gas flow (L.day^{-1}), R is the gas constant (atm$\times$m$^3\times$mole$^{-1}\times$K^{-1}), T is the temperature (K), and H is Henry's law constant (atm$\times$m$^3\times$mole^{-1}) at 20°C, taken from Ten Hulscher *et al* (1992). When the volatilizing chemical is trapped on Tenax columns, C_W can be calculated from the cumulative trapped amounts, Q^{TENAX}, using:

$$C_W = \frac{Q_{t=0} - Q^{TENAX}}{V_w} \tag{2}$$

in which $Q_{t=0}$ (μg) is the amount of chemical initially present in the system. In case of a DOC containing medium at sorption equilibrium, the volatilization process induces desorption. If the rate of gas purging is sufficiently slow so that equilibrium between DOC and water is maintained, and the DOC concentration remains constant during purging, then the gas purge induced HOC removal from the medium is described by (Hasset and Millicic, 1985):

$$dC_T/dt = -\frac{k_v}{1 + (DOC \times K_{DOC})} \times C_T \tag{3}$$

in which C_T (μg/l) is the sum of aqueous and DOC associated HOC concentrations, DOC (kg/l) is the organic carbon concentration and K_{DOC} (L/kg) is the exudate-water association constant on an organic carbon basis, as determined by gas purging. Because k_v and *DOC* are known, Equation 3 allows the calculation of K_{DOC} values. Parameters were estimated by non linear least squares regression.

RESULTS AND DISCUSSION

Culture characteristics - The characteristics of the culture and medium are presented in Table 1. The organic carbon fraction, f_{oc}, of 0.56 agrees very well with literature values for algal cells (Jørgensen *et al*, 1991). Further, it appears that the DOC concentration is not significantly affected by purging.

In the medium a considerable number of particles was counted after centrifugation. Direct microscopic inspection revealed that no algal cells were present. Bacteria are known to grow efficiently on algal exudates (Lignell *et al*, 1993; Hambsch *et al* 1993) and may have contaminated the DOC-pool in the medium. However, as f_{oc} was determined as (TOC-DOC)/DW, an overestimation of the DOC fraction by bacterial contamination would have resulted in an unrealistic low f_{oc} value, which is not observed. Furthermore, assuming a bacterial cell size of 1 μm (Jørgensen *et al*, 1991) and a biovolume to biomass conversion factor of 0.22 g C cm^{-3} (Bratbak and Dundas, 1984), 0.28×10^{9} bacterial cells$\times L^{-1}$ (Table 1) could account only for a negligible 0.5% of the measured DOC concentration. Even if the counts would relate to planktonic particles, (contradicting the microscopic inspections), the particles could account for up to 25% of the observed DOC concentration, still leaving dissolved exudates as the dominant carbon pool.

TABLE 1 Characteristics of the *Selenastrum capricornutum* culture and medium.

object/age[a]	Cells $\times 10^{9}L^{-1}$	DW[b] mg/L	TOC[c] mg/L	DOC[d] mg/L	f_{oc}[e] -
culture, day 3	1.63	18.7±0.47	16.8±0.3	-	0.56±0.02
medium, day 3	0.28	-	-	6.40±0.18	-
medium, day 17	0.49	-	-	6.20±0.21	-

[a]Days after HOC spike; [b]Dry weight after 0.45 μm filtration; [c]Total organic carbon in complete culture; [d]Total organic carbon after centrifugation; [e]Organic carbon fraction of cells: f_{oc}=(TOC-DOC)/DW.

Bioconcentration - The *BCF* values after three days of uptake are presented in Figure 1A. The regression equation has a slope of 0.63 (r^2=0.895, p=0.015). The less-than-unity slope reflects a significant effect of DOC or nonequilibrium due to the rather short uptake time.

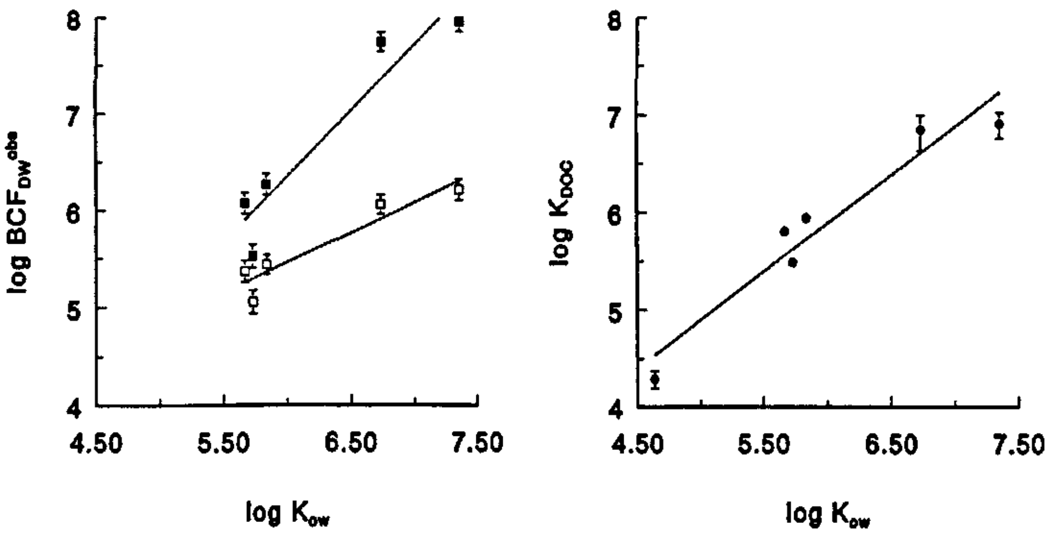


Fig.1. A: LogBCF_{DW}^{obs} vs. logK_{ow} and linear regression line after three days of uptake. Error bars represent 1 s.d. Observed: --□--. Corrected for HOC-DOC association (Equation 4):--■--.
B: LogK_{DOC} vs. logK_{ow} and regression line (--●--) for HOC binding to extracellular carbon. Error bars represent 1 s.d.

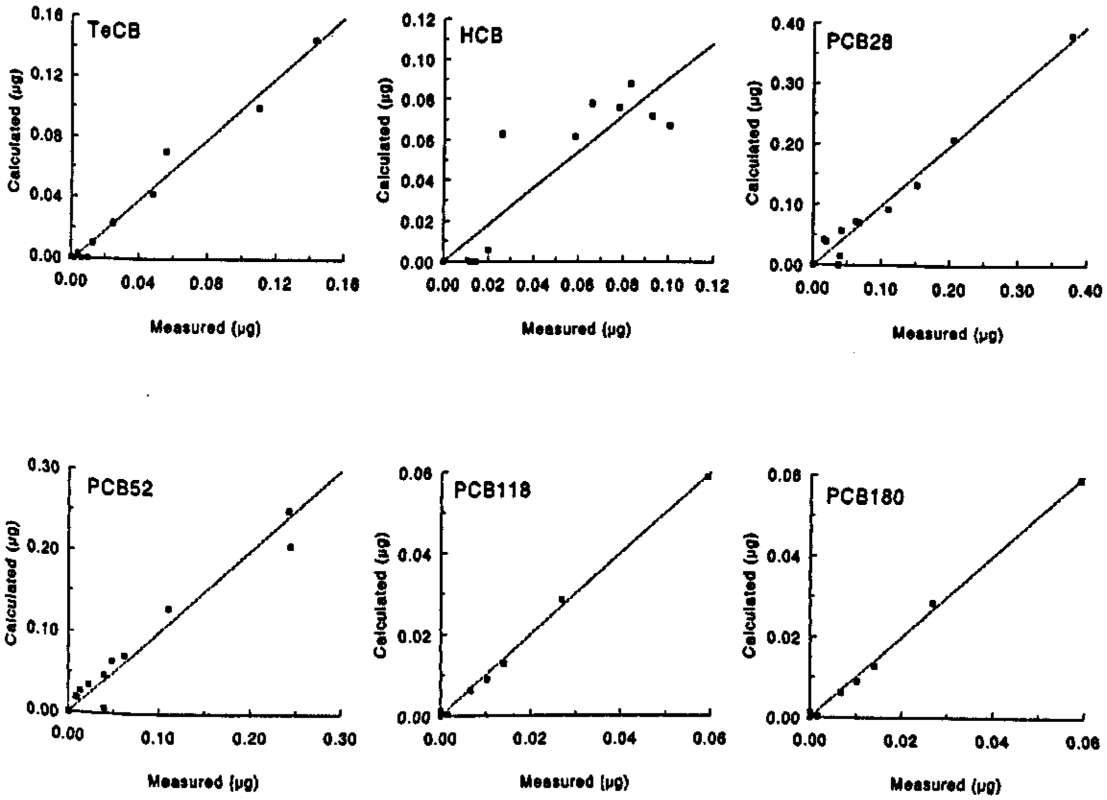


Fig.2. Comparison of calculated (Equation 3, first order equilibrium binding model) and measured HOC amounts purged from an exudate containing medium in varying time periods, for TeCB (r^2=0.981), HCB (r^2=0.770), PCB28 (r^2=0.971), PCB52 (r^2=0.955) and PCB180 (r^2=0.997). The ideal 1:1 lines are shown for guidance.

Association with exudates - Values for K_{DOC} were calculated from the sequential amounts trapped in the tenax columns as a function of time. The first order equilibrium model as condensed in the equations 1 - 3 fitted the gas purge data well. The quality of fit is illustrated in Figure 2, by comparing the calculated model predictions with the measured values. The ideal 1:1 line is shown for guidance. For HCB the model was significant at p=0.03 (F-test on residual errors, n=12, r^2=0.77). For the other test compounds significance levels were excellent ($5\times10^{-7}<p<8\times10^{-5}$ and $0.955<r^2<0.997$). The logK_{DOC} values determined by gas purging are presented in Figure 1B and Table 2. The logK_{DOC} on logK_{ow} regression equation (Figure 1B) has a slope of 1±0.14 and an intersect of -0.055 (r^2=0.927, p=0.002, n=6). So the logK_{DOC} values are almost equal to logK_{ow} which means that the extracellular carbon in *Selenastrum* cultures binds PCBs very effectively. Note that the results of the regression calculations and the magnitude of the slopes, depend on the selected K_{ow} values, which may vary half a log unit among literature sources. Nevertheless, the high values directly prove that HOC-DOC associations are important in the interpretation of phytoplankton bioconcentration data in the field or the laboratory, and therefore strongly support the qualitative discussions on possible DOC effects in earlier reports (eg., Richer and Peters, 1993; Stange and Swackhamer, 1994; Sijm *et al*, 1995; Koelmans *et al*, submitted). The high K_{DOC} values are consistent with those reported by Lara *et al* (1989), who measured association constants for exudates from brown algae. From their data we calculated the following linear relationship: $\log K_{DOC}$= $1.89\log K_{ow}$+0.541 (n=19, r^2=0.913). Thus their association constants are even higher than those measured in the current study. Exudates are a poorly defined mixture of metabolic intermediates (glycollic acids, polysaccharides) and metabolic endproducts (carbohydrates, peptides, enzymes, vitamins and toxins) (Fogg, 1971). Further, in batch bioconcentration experiments small fractions of bacterial carbon may contribute to the extracellular carbon pool. The variability of the composition of the extracellular carbon suggests a high variability of K_{DOC}, which explains the difference between the results of Lara *et al* and ours. The extracellular organic molecules mentioned above are soluble compounds and may be expected to be rather hydrophillic. It should be noted that the probably hydrophillic nature of exudates is not consistent with the observed high K_{DOC} values found, which may be subject of further study.

The effect of HOC-DOC association on the bioavailability may be quantified with the product DOC×K_{DOC} as in Equation 4 (Gschwend and Wu, 1985; Koelmans *et al*, submitted):

$$BCF^{observed}=\frac{BCF^{true}}{1+DOC\times K_{DOC}} \tag{4}$$

If this product for a certain HOC-DOC association is larger than one, bioavailability is reduced. DOC×K_{DOC} products for our experimental conditions are listed in Table 2. It appears that for HCB and the PCBs, exudation significantly reduced the bioavailability for uptake. Table 2 also shows the percentage of the dissolved chemical which is not associated with DOC, for DOC=6.4 mg/l (current experiments) and a hypothetical DOC of 0.5 mg/l. For PCB180, 98% of the dissolved concentration is as soluble PCB-DOC complex. We conclude that for the more hydrophobic PCB congeners, HOC-DOC associations in natural waters may become relevant already at DOC levels of 0.5 mg/L.

TABLE 2: HOC-DOC association constants K_{DOC} (kg/L) for Selenastrum exudates at 20°C.

	$\log K_{ow}$	$\log K_{DOC}$	DOC×K_{DOC}[c] kg/L×L/kg	Available at 6.4 mg/L, %	Available at 0.5 mg/L, %
TeCB	4.635[a]	4.29	0.125	89	99
HCB	5.731[a]	5.49	1.96	34	87
PCB28	5.67[b]	5.81	4.12	20	76
PCB52	5.84[b]	5.95	5.78	15	69
PCB118	6.74[b]	6.86	46.7	2	22
PCB180	7.36[b]	6.92	53.2	2	19

[a]From De Bruijn *et al* (1989); [b]From Hawker and Connell (1988); [c]DOC=6.4×10^{-6}kg/L

Using Equation 4, and the K_{DOC} values in Table 2, the observed *BCF* values can be converted to pure algae/water partition coefficients. The corrected values are plotted in Figure 1A. The regression equation now has a high slope of 1.37 (r^2=0.897, p=0.014). Note that the uncertainty in the corrected *BCF*s is relatively high due to error propagation. The high slopes suggest that algae are more efficient HOC carriers than pure octanole.

CONCLUSIONS AND IMPLICATIONS

In this study high association constants were found for binding of chlorobenzenes and polychlorobiphenyls to *Selenastrum capricornutum* exudates. The constants were more or less equal to

K_{ow}, suggesting rapid partitioning as the association mechanism. The high constants imply that exudation significantly affects the bioavalability of hydrophobic chemicals. This also implies that binding to DOC should be accounted for in the interpretation of laboratory HOC bioconcentration data. As the cycling of detrital carbon in eutrophic waters is highly dynamic, and dissolved carbon fluxes may be considerable, binding to DOC should be accounted for in HOC fate and food chain accumulation modelling. If the high constants are a general feature of HOC-exudate associations, exudation and detrivory may resemble important HOC loss resp. uptake mechanisms for planktonic species.

ACKNOWLEDGMENT

The technical assistance of Frits Gillissen is gratefully acknowledged.

REFERENCES

Bratbak, G. and Dundas I. (1984). Bacterial dry matter content and biomass estimations. *Appl. Environ. Microbiol.*, 48:755-757.

De Bruijn, J., Busser, F., Seinen, W. and Hermens, J. (1989). Determination of octanol/water partition coefficients for hydrophobic organic chemicals with the "slow-stirring" method. *Environ. Toxicol. Chem.*, 8:499-512.

Fogg, G.E. (1971). Extracellular products of algae in freshwater. *Egebnisse der Limnologie*, Heft 5:1-25.

Gschwend, P.M. and Wu, S.C. (1985). On the constancy of sediment-water partition coefficients of hydrophobic organic pollutants. *Environ. Sci. Technol.* 19: 90-96.

Hambsch, B., Werner, P., Mäckle, H. and Frimmel, F.H. (1993). Degradation of algal exudates by mixed bacterial biocenosis. *Wat. Sci. Tech.*, 27:421-429.

Hasset, J.P. and Millicic, E. (1985). Determination of equilibrium and rate constants for binding of a polychlorinated biphenyl congener by dissolved humic substances. *Environ. Sci. Technol.*, 19:638-643.

Hawker, D.W. and Connell, D.W. (1988). Octanol-water partition coefficients of polychlorinated biphenyl congeners. *Environ. Sci. Technol.*, 22:382-387.

Jørgensen, S.E., Nielsen, S.N. and Jørgensen L.A. (1991). *Handbook of ecological parameters and ecotoxicology*. Elsevier, Amsterdam.

Kilham, S., Goulden, C.E., Kreeger, D. and Herrera, L. (1995). COMBO A defined freshwater culture medium for algae and zooplankton. *ASLO 1995 meeting, Book of Abstracts*

Koelmans, A.A., Sánchez Jiménez, C. and Lijklema, L. (1993). Sorption of chlorobenzenes to mineralizing phytoplankton, *Environ. Toxicol. Chem.*, 12:1425-1439.

Koelmans, A.A., Anzion, S.A. and Lijklema, L. (1995). Dynamics of organic micropollutant biosorption to cyanobacteria and detritus. *Environ. Sci. Technol.*, **29**:933-940.

Lara, L.J., Wiencke, C. and Ernst, W. (1989). Association between exudates of brown algae and polychlorinated biphenyls. *J. Appl. Phycol.*, **1**:267-270.

Lignell, R., Heiskanen, A.-S., Kuosa, H., Gundersen, K., Kuuppo-Leinikki, P., Pajuniemi, R. and Uitto, A. (1993). Fate of a phytoplankton spring bloom: Sedimentation and carbon flow in the planktonic food web in the northern Baltic. *Mar. Ecol. Prog. Ser.*, **94**:239-252.

Richer, G. and Peters, R.H. (1993). Determinants of the short-term dynamics of PCB uptake by plankton. *Environ. Toxicol. Chem.*, **12**:207-218.

Sijm, D.T.H.M., Middelkoop, J. and Vrisekoop, K. (1995). Algal density dependent bioconcentration factors of hydrophobic chemicals. *Chemosphere*, **31**:4001-4012.

Stange, K. and Swackhamer, D.L. (1994). Factors affecting phytoplankton species-specific differences in accumulation of 40 polychlorinated biphenyls (PCBs). *Environ. Toxicol. Chem.*, **13**:1849-1860.

Swackhamer, D.L. and Skoglund, R.S. (1993). Bioaccumulation by algae: kinetics versus equilibrium. *Environ. Toxicol. Chem.*, **12**:831-838.

Ten Hulscher, Th.E.M., Van der Velde, L.E. and Bruggeman, W.A. (1992). Temperature dependence of Henry's Law constants for selected chlorobenzenes, polychlorinated biphenyls and polycyclic aromatic hydrocarbons. *Environ. Toxicol. Chem.*, **11**:1595-1603.

EFFECTS

GROWTH KINETICS OF SOME CYANOBACTERIA

L.R.Mur
ARISE, University of Amsterdam

ABSTRACT

Each year diatoms start and are followed by populations of chrysophytes. In summertime green algae or cyanobacteria become the dominant group. In autumn a second population of diatoms is found sometimes. These phenomena have been described by generations of limnologists. Sometimes we think that we understand the behaviour of the different populations, which means that their wax and wane fit with the picture built from ecophysiology or growth kinetics. However mostly our knowledge is too limited to understand or to predict the behaviour of algal populations. This is not strange when we realise that of the diatoms only 1.000.000 different species have been described at this moment. Each species with its own properties. Not all these species are important for the Dutch situation., but plankton lists from one sample containing over 100 species are normal. Realising that at least 10-20 abiotic ecological factors can play a decisive role in competition, it becomes clear that the possibilities to understand the behaviour of plankton succession from ecophysiological information is just starting. When we incorporate also the biotic components the situation becomes unmanageable.
The only possibility to reach a manageable situation is to form groups of species of which is expected that their niches are quite comparable. Often these groups correspond with the systematical groups for example: diatoms are always found in springtime because they need silicon which is only available in that period and they are psychrophilic which makes it possible that they start growing in early spring. Another group is the Chlorococcales, in eutrophic waters many species of Scenedesmus , Crucigenia and Pediastrum can be found together. In theory they must all have different niches, but in practice they seem to be found all together in dynamic eutrophic waters.

Not only the knowledge of the growth kinetics of the different groups is limited but also the interpretation of the ecological factors. How must we interpret a nutrient limitation. We often interpret a nutrient limitation as a constant factor over a period of time, as we do in continuous cultures, but how important are nutrient pulses from the sediment, how important is the competition for ammonia between nitrifying bacteria and phytoplankton species. This situation is even more complicated when light is considered as a substrate. At light limitation the organisms can increase their light harvesting complexes which increases their affinity for light. However, how does the organism react under light saturating conditions, when it has to protect itself against a constant bombardment of photons. When the bombardment is not to intensive it can protect itself by light dissipation , which we know as fluorescence, or heat dissipation. higher light conditions photodamage becomes a danger. It is clear that the effects of this process are intensity and time dependent.

What part of this information must we incorporate in our studies of the growth kinetics. In all models used to describe phytoplankton growth practical choices have been made mostly based on the behaviour and distribution in nature, sometimes on the base of eocphysiological information.

The knowledge concerning the growth kinetics of cyanobacteria becomes more and more detailed.. Here a number of examples are given in which the ecological distribution and ecophysiological information are combined to a complete picture of their behaviour in natural waters.
The most complete information is known of the group of Oscillatoria species. They have gasvesicles which means that their loss factor by sedimentation is low but their vertical migration is very limited by the absence of colony formation.. So the whole population is homogeneously divided over the water column. In highly eutrophic light limited situations Oscillatoria wins the competition by its highly efficient phototrophic growth. They form a dense population which in normal Dutch winter survives the cold period in relatively large numbers, resulting in a high start population in the next growing season. Their affinity for phosphorus is relatively high which suggests that they will outcompete other organisms also in phosphorus limited situations. However when the amount of available phosphorus becomes too low the maximal population decreases and the average light climate in the water column becomes favourable for other organisms or even so high that photoinhibition damages the population. Modelling the population dynamics of Oscillatoria in shallow lakes can largely be based on the influence of the light climate on the growth of this organism. The same is possible with the behaviour of the metalimnetic strains of the phycoerythrin containing Oscillatoria species. The organisms have a combination of antennae pigments which make them highly adapted to the light climate in the metalimnetic zone. They are also very light sensitive and the organisms disappear when the lake becomes really oligotrophic.
It is surprising to find Prochlorothrix hollandica in highly eutrophic peat lakes. These lakes look very suitable for Oscillatoria. If the niche of Prochlorothix hollandica is comparable with the niches of the Oscillatoria species, why Prochlorothrix is not found in the other eutrophic shallow lakes, like the 'Randmeren'? The most obvious explanation might be that the special combination of photosynthetic pigments and prokaryotic cell structure gives this organism the best combination of properties to live in the brownish water of the peat lakes. However very recent data learn that this organism can store much more phosphorus in its cells than known from the Oscillatoria species. This makes it attractive to develop a kinetical model for this species in which not light, but a dynamic phosphorus supply is the steering factor although there is hardly any knowledge of the phosphorus dynamics in this type of lakes.

A totall different type of growth kinetics is found in Microcystis aeruginosa. In contradiction to the Oscillatoria species this organism has a clear life cycle during the year. The organism is overwintering on the bottom of the lake. In anaerobic sediments it can survive by fermentation. The stored glycogen is fermented to ethanol, acetate, CO_2 and H_2, delivering enough energy to survive and perhaps to grow. By the loss of glycogen its density diminishes, which makes that the organisms become buoyant and come back in the epilimnetic zone. In spring they form populations which are unicellular or consist of small colonies of only some cells. The size of the particles makes that the organisms do not migrate through the water column and are easy to be grazed. Later in the summer the colonies grow and the organisms become difficult to graze, start migrating through the epilimnion and form surface scums. In autumn time their high photosynthesis and low growth rate makes them heavy and they sink back to the sediment. The scum formation and the potential toxicity of this species makes it extremely important for watermanagement. However how to describe the population dynamics of this species in such a way that it can be used for algal modelling. Integrating all parts of the life cycle makes the model quite complicated. What are the critical parts of its life cycle, is it the anaerobic sediment in which they can survive, how important are the loss factors in springtime, how can we modulate the growth in summertime, during which they migrate through the light gradient. All these factors make that no good dynamic models are available concerning the growth of Microcystis. All models only use the information from the field ecology, like the relation between blooming and temperature, or scumformation and temperature stability.

Coming to a conclusion it must be stated that algal growth kinetics are extremely diverse. In the relatively small group of cyanobacteria many different strategies for survival , overwintering and photosynthetic growth are found. The ecophysiologist will continue to unravel these strategies, it is the challenge of the ecological modeller to use this information in a proper way.

THE E NUMBERS OF EUTROPHICATION - ERRORS, ECOSYSTEM EFFECTS, ECONOMICS, EVENTUALITIES AND ENVIRONMENT

Brian Moss

School of Biological Sciences, University of Liverpool
Derby Building, Liverpool L69 3BX, UK

ABSTRACT

Accumulated experience is a precious commodity. Fifty years ago, Aldo Leopold succinctly expressed many of the principles of land management that contribute to minimising eutrophication. Lessons from his essay 'Odyssey' can be drawn and applied our current understanding of eutrophication. They include the importance of nitrogen vis a vis phosphorus, the relevance of a whole system treatment of the problem, the costs of the problem, the value of intact ecosystems in providing goods and services, and the future use of land in mitigating the problem on a world scale.

KEYWORDS

Eutrophication; nitrogen; phosphorus; Ado Leopold; nutrient cycling; economics

INTRODUCTION

Aldo Leopold, Professor of Wildlife Management at the University of Wisconsin, until his untimely death fighting a neighbour's grass fire in 1948 (Gibbons 1981), understood eutrophication and its consequences long before the several tens of thousands of subsequent references to the subject were published, or the word was even in wide circulation. He wrote a short essay (Leopold 1949), formerly for Audubon Magazine, called *'Odyssey'*, about an atom, X, which becomes weathered from a rock, passes through multiple cycles in the prarie ecosystem of mid-western America, enters the atmosphere through a prairie fire, is rained back upon the land, and reaches a wetland ecosystem in a cottonwood tree then a beaver, and its carcass. The carcass becomes silt, and X feeds a crayfish, a raccoon and an Indian and eventually reaches the sea. It is the now familiar story of a biogeochemical pathway in which the land ecosystem uses many subcycles to retain the atom in circulation and therefore continually available to the system. Eventually the atom is lost to a sink on the sea bed but not before it has served many roles, over a long period in the terrestrial ecosystem. Leopold understood a rough balance between supply and loss to geological sinks...*'For every atom lost to the sea, the prairie pulls another out of the decaying rocks. The only certain truth is that its creatures must suck hard, live fast, and die often, lest its losses exceed its gains'*

A second, sister atom Y finds itself making a shorter and faster odyssey, displaced from a prairie converted to arable agriculture and soon entering the watercourse, a lost atom *'that once grew pasque-flowers to greet the returning plovers now (lying) inert, confused, imprisoned in oily sludge'*, which had accumulated from the erosion of the land and the disgorging of the sewers. It is a comprehensive account of the eutrophication that has followed development and agriculture.

In less than 1400 words, half a century, ago Leopold distilled a lifetime of observation and offers us insights that may have become occluded in the welter of publications that an extravagant age seems to feel necessary. This article will pick up some of these insights and I believe that this is a fitting way to celebrate the long and productive career of Professsor Lijklema. It is easy to forget, now that many literature searches are computer-based and assume that science began in about 1981, that much of what appears to be new is, like atom X, often only the newly recycled understanding of our older colleagues.

The literature on eutrophication seems prone to alliteration. We have had books on its 'Causes, Consequences and Correctives' (National Academy of Sciences 1969) and its 'Expectations, Experiences and Extrapolations'(Sas 1989). In this tradition and with the inspiration of the E numbers which designate approved food additives in the European Community, but which many people take as indications to beware

(Hanssen & Marsden 1984), I will discuss some possible errors, ecosystem effects, economic, educational, eventual and environmental issues, which are tucked into Leopold's essay and philosophy and some of which have been concerns in Professor Lijklema's career.

ERRORS

Leopold's essay does not name atoms X and Y, and this might remind us that eutrophication theoretically involves a range of elements. Recent work in the ocean has turned attention to iron as a potential eutrophicating nutrient, the functioning of some highly infertile lakes, such as Loch Ness, suggests that ingress of carbon might stimulate production through the bacterial-protozoon pathways in a lake where deep mixing limits phytoplankton photosynthesis even in summer (Jones et al 1996). The growth rate of algae might be limited by a variety of factors quite apart from nitrogen and phosphorus, and the rate at which their biomass accumulates is frequently determined by grazing. Where the standing crop is concerned, however, I would find it hard to accept that we have been seriously wrong in the emphasis that has been given to nitrogen and phosphorus in freshwater systems. On the other hand we may have emphasised phosphorus too much and nitrogen too little as the prime driver, though always, increases in both will have been necessary to give severe, if not minor problems. Leopold hints that it is nitrogen that he has in mind as much as any element as the traveller of his odyssey - *'Thus the prairie savings bank took in more nitrogen from its legumes than it paid out to its fires'.*

There is evidence that even in some deep lakes, mechanisms may exist where phosphorus is allowed to build up in the water and production is controlled by nitrogen. Such lakes have strong summer stratification, migratory dinoflagellates or cyanophytes and long retention times because they are fed and drained largely by ground water. They include some of the glacial kettle hole lakes of the west midlands of the UK (Reynolds 1979, Moss et al 1994), where in landscapes of only moderate agricultural intensification, and phosphorus-poor groundwater, lake concentrations of several hundreds of micrograms per litre of TP are normal. Furthermore, palaeolimnological studies of some of them now show that these lakes had abundant cyanophytes at least as long as 6000 years ago (McGowan 1997). The mechanism of phosphorus accumulation may comprise low washout, coupled with transfer in summer of phosphorus by migratory algae from the epilimnion to the hypolimnion, saturation of the sediments, and low nitrogen input due to denitrification in the wet meadows and reedswamps that encircle them (Moss et al 1997). There has been some recent eutrophication superimposed on their naturally eutrophic state and this must have been driven by nitrogen.

Such lakes may not be unusual among the tens of thousands of pothole or kettle hole lakes that dot the glacial plains of the northern hemisphere. Some lakes, surrounded by undisturbed boreal forest on the glacial plains of Alberta, Canada, for example, have dense cyanophyte blooms. Tropical African lakes may also be more likely to be nitrogen rather than phosphorus limited (Talling & Talling 1965, Moss 1969). This may be a function of high catchment temperatures, favouring mineralisation of organic matter in soils, vigorous denitrification in warm wetlands and isolation of many tropical water bodies in the dry season, leading to sediment recycling of phosphorus but preventing ingress of new supplies of nitrogen from the catchment.

Closer to home may be a greater importance of nitrogen in driving the eutrophication of macrophyte dominated shallow lakes. Professor Lijklema has contributed (1993,1994) much understanding to the recycling of phosphorus from sediments, particularly in shallow lakes. It is clear that phosphorus escapes from sediments much more readily than previously assumed and our thinking otherwise may simply have been due to the accident that the early, classic studies of lake restoration from eutrophication involved big lakes in rocky bsins with vigorous flow through of water. Lijklema (1994) points out that enhanced eutrophication favours a shift towards nitrogen limitation. This is partly because excretally derived nutrient sources have low N:P ratios and because organic loads to sediments in eutrophicated lakes stimulate deoxygenation and promote denitrification.

What might be more controversial, however, is the possibility that in shallow lakes, especially those dominated in their pristine state by macrophytes, eutrophication has been driven largely by nitrogen inputs. We might expect that in shallow lakes, phosphorus has always been relatively abundant, whilst nitrogen, because of the readiness by which it is denitrified by sediment communities, might have been relatively scarcer and potentially limiting.The implication of this is that phosphorus control might be futile as a restoration measure for shallow lakes. The evidence for such a hypothesis is not yet clear but there are some interesting indications.

First, the summer water of shallow lakes dominated by macrophytes is usually highly deficient in nitrogen though phosphorus is often abundant (Ozimek et al 1990; Moss et al 1997); secondly, in experiments with mesocosms, extant concentrations or additions of nitrogen disappear very rapidly but those of phosphorus less so (Beklioglu & Moss 1996a,b, D.Stephen, unpublished data); thirdly, although the processes by which macrophytes disappear in eutrophicated lakes are complex and involve switch mechanisms in addition to increased nutrient loading, the diversity of submerged plants is linked to nitrogen loading as measured by both inflow and lake concentrations.Many factors may determine the number of species but in examples culled from the literature (Fig.1), there is an envelope constraining high diversity to low total n concentrations and limiting diversity at high concentrations. There is a much less clear trend with phosphorus concentration(Fig.2).

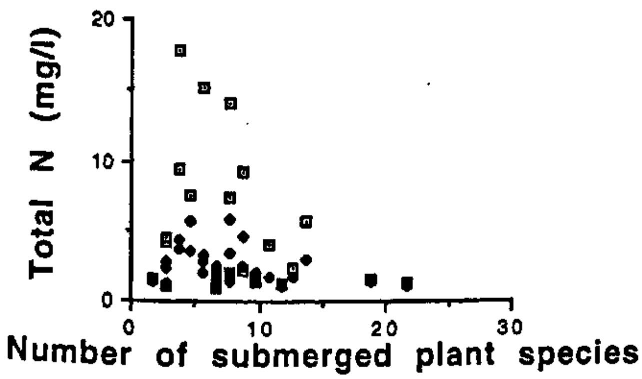


Fig.1 Relationship between mean annual total nitrogen concentration in lake and inflow waters with number of submerged plant species in a group of British lakes. Data from B.Moss, P.J. Johnes & G.Phillips (unpublished) and the public registers of the Environment Agency. Open squares are inflow values, solid diamonds are lake values.

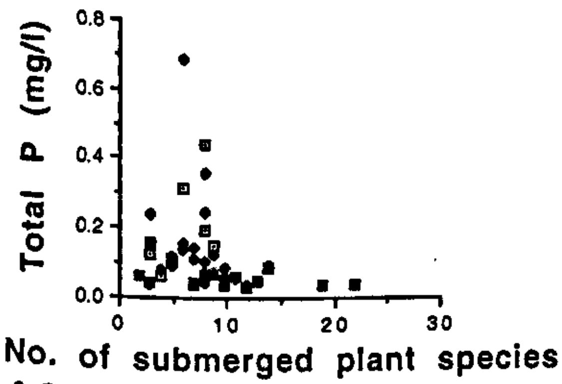


Fig.2. Relationship between number of plant species and total phosphorus concentrations in lake (open symbols) and inflow (closed symbols) waters from a group of British lakes.

The problem with these data however is that they often derive from lakes that have been severely affected by eutrophication or have been recently partly restored from it.There may be long lag effects in which phosphorus is released from sediments at abnormally high rates (Lijklema 1994), and in which the importance of nitrogen is temporarily over-emphasised. It would be most useful to have a set of lakes unaffected by human activities to examine the relationships between nitrogen and macrophytes but this is clearly not possible in the developed world, except in remote mountain aeas where the conventional view of the key importance of phosphorus is likely to be correct.

However, by using nutrient export modelling that has been fully calibrated and validated against real data, and historic records of land usage, stock numbers and population, it is possible to reconstruct something of former nutrient regimes (Johnes 1996, Moss et al 1996). When this was done for ten catchments in the UK (Johnes et al 1997), former N:P ratios were found to be similar to current ones in generally upland or rolling catchments with fast run off of water but not in a flat lowland catchment, with now-eutrophicated shallow lakes where the N:P ratio had nearly doubled in about fifty years, suggesting possible former N limitation. Studies of a group of lakes widely distributed in the UK and containing many different types (Moss et al 1996) (Fig 3) showed that total nitrogen concentrations have increased by a factor of about five from the hindcasted state, but total phosphorus concentrations have increased by about twelve-fold.This is not helpful in supporting the hypothesis because it merely suggests that nitrogen has become relatively scarcer as the eutrophication problem has developed.

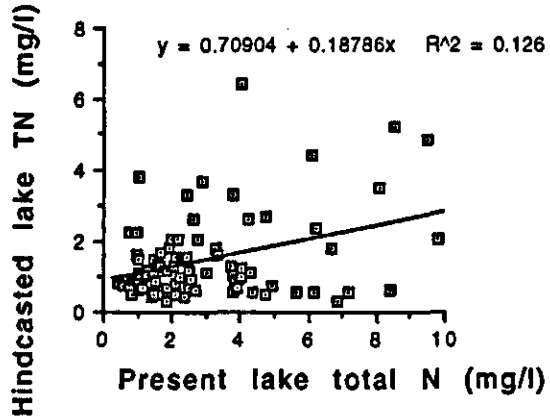


Fig.3 Relationship between present and hindcasted total nitrogen concentrations in a group of British lakes. Data from Moss et al 1996.

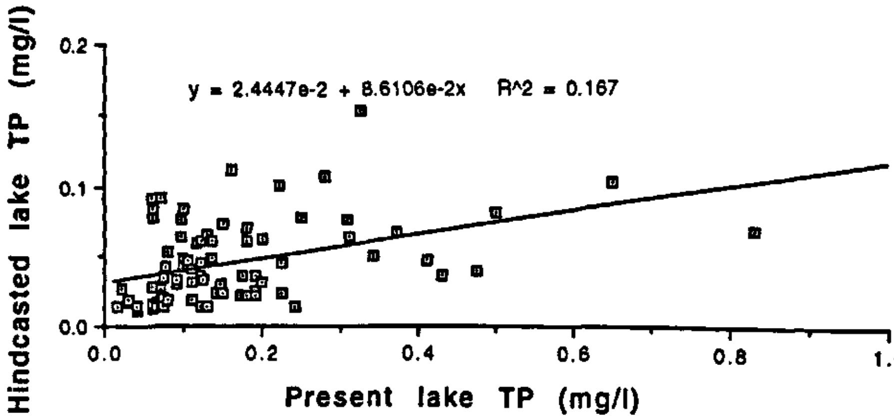


Fig.4 Relationship between present and hindcasted total phosphorus concentrations in a group of British lakes. Data from Moss et al 1996.

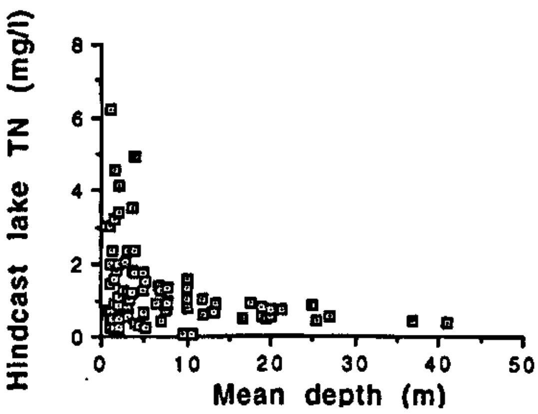


Fig.5 Relationship between hindcasted total N concentration and mean depth in a group of British lakes. Data from Moss et al 1996

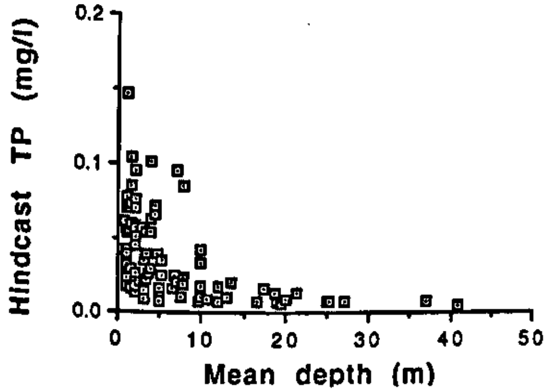


Fig.6 Relationship between hindcasted total P concentrations and mean depth in a group of British lakes. Data from Moss et al 1996.

It might be expected that nitrogen and phosphorus concentrations would be greater in shallow lakes, even in the past, because of the general differences in catchments expected of deep and shallow basins. This proves to be the case in the hindcasting (Figs 5,6). However, when the hindcasted N:P ratios are examined in relation to depth of lake, there appears to be an unexpected positive relationship such that former N:P ratios in shallow lakes were low and therefore that nitrogen was more likely to have been limiting than in deeper lakes (Fig.7). This suggests that increases in nitrogen may have been more crucial in these than in deeper lakes.

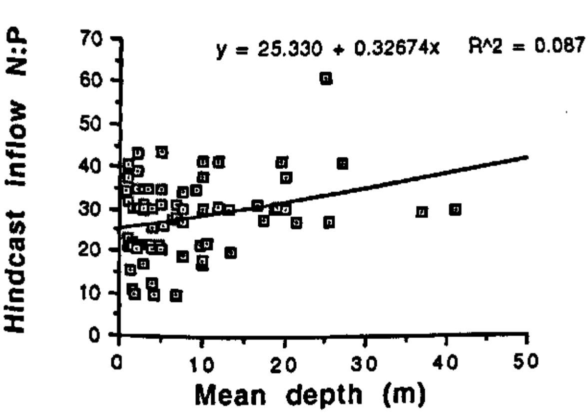


Fig.7 Relationship between hindcasted inflow total N to total P ratios in a group of British lakes. Data from Moss et al 1996.

The data are indicative rather than conclusive but suggest that in the future, in the lowlands, we shall have to take nitrogen control more seriously. Conventional views are that nitrogen control is futile because of the existence of nitrogen fixation (Schindler 1977). However, fixation is relatively slow, otherwise situations of nitrogen limitation would not develop. The recycling of fixed nitrogen from the fixers to other organisms takes time. Nitrogen control at sewage treatment works is expensive but technically feasible and covered under the EC Urban Waste Water Treatment Directive; control of agricultural and husbandry sources is much less easy. Some progress has been made using buffer zones but the ultimate solution must lie in a less intensive use of land, with lower fertilisation rates and less intensive cultivation. The projected performance of a variety of strategies for reducing diffuse sources of both N and P is shown in Table 1(from Johnes 1996). It appears that buffer zones offer much less control than reduction in the intensity of the farming. The EC Nitrate Directive, through the designation of nitrate sensitive areas (NSA) offers greatest potential but in the UK, at least, it has been used only to control nitrate concentrations in drinking water supplies, largely in groundwater. The critical concentration of 50 mgl^{-1} (11.3 mg N l^{-1}) designated for water supplies in the Nitrates Directive is far too high to have much impact for freshwater conservation and management of river weeds. Even so one UK House of Lords Select Committee calculated that reducing concentrations to this level in all waters would reduce UK arable production by 33% and cost £4billion (at 1987 prices) (Barnden 1993). The assumption that the agricultural industry must be compenated for desisting from damaging practices is beginning to look outmoded however and a severing of the close link between agriculture and the provisions of the food processing industry might mean that a reduction in arable agriculture would not be so disastrous as its lobbyists intend it to sound. Lijklema (1994) argues that, for various reasons, applications of nitrogen fertilizer are likely to decrease in Europe but to increase elsewhere.

Table 1. Effects of a variety of management options on the projected mid 21st century concentrations of total N and total P in waters discharging to the River Windrush in the Cotswold Hills, UK and to a small Lake, Slapton Ley, in South Devon, UK. NSA is Nitrate Sensitive Area. Redistribution of land use would mean relocation of certain activities to shallower slopes. Basin specific strategies involve reduced intensification on a targeted basis. Values are percentage change from values in the late 1980s. Based on Johnes (1996).

Option	Slapton Ley		River Windrush	
	N	P	N	P
Current trend	+93	+61	+28	+7
Reduce fertiliser appn by 20%	-8	-10	-10	0
Basic NSA scheme	0	0	-4	-1
Premium NSA scheme	-8	-17	-20	-22
Basin specific strategy	-15	-17	-18	-17
Redistribution of land use	-8	-11		
Riparian strip (50m)			-3	-2

ECOSYSTEM EFFECTS

Aldo Leopold had a profound sense of the links between processes. It was not some romantic concept of the balance of nature, but a profound understanding that actions have chains of consequences. The fate of atom Y epitomises this. Eutrophication is not the simple addition of an undesirable substance, it is the profound alteration of entire systems. In eutrophication research, the ecosystem consequences have generally been considered, perhaps because fish and fisheries, lying at the far pole of the food web, have always been prominent in considerations of the use of freshwaters. There are strong interactions between nutrient loading and predation, the other major determinant of freshwater communities. In ecosystem consequences, the effects of eutrophication are perhaps more obvious but probably no more consequential than the addition of other polluting substances - heavy metals, trace organics such as industrial by products and pesticides. Yet the lessons learned from a long history of research into eutrophication have not been applied to other forms of pollution.

Emphasis here is on effects on specific species, or their internal metabolism rather than widely ramifying effects of damage to the system as a whole. There is thus now a considerable emphasis on ecotoxicology, the thrust of which appears (to me at least) to be that if sophisticated methods of detecting toxins can be found, the problem is solved. Research emphases have gravitated more towards the test tube than the environment and the philosophy, of which Leopold would have approved , that ultimately the solution to toxic pollution problems is never to allow toxin release into the environment, becomes obscured by the sophisticated fashions of molecular detection.'*We abuse land because we regard it as a commodity belonging to us. When we see land as a community to which we belong, we may begin to use it with love and respect*' (Leopold 1949, Foreword).

Such non-system considerations underly the setting of permitted discharges by emission (end of pipe) standards. Lijklema (1995a) has discussed the problems in setting these because they take no note of the nature of the receiving ecosystem. Similarly the setting of 'immission standards' (receiving water quality standards) also has problems because of the natural variability in background concentrations, and the variability within a water body and through diurnal and seasonal cycles. In all pollution control, it is desirable that standards are relevant to the local circumstances of a water body. There is little sense in creating the same heavy metal standard for a spring that emerges below a natural ore body and a stream draining sandstone, or phosphorus standard for a lowland lake in glacial drift deposits and an upland one on granite. Most systems of determining water quality assessment are arbitrary and simply compare the present status of spatially distributed locations. It would be much better to compare the present state with a past reference or baseline state, which need not be the pristine (absence of people) state but some baseline that is appropriate to the location and leaves the catchment in an indefinitely sustainable condition. Such schemes are most advanced for characteristics related to stream pollution (e.g. the RIVPACS scheme, Wright 1995), structural modification of rivers (the Petersen (1992)

RCE scheme) and nutrient loading changes on lakes (Moss, Johnes & Phillips 1996, 1997). Inherent in their thinking is the comparison made by Leopold for atoms X and Y for before and after the prairie was ploughed.

ECONOMICS

'The old prairie lived by the diversity of its plants and animals, all of which were useful because the sum totl of their co-operations and competitions achieved continuity. But the wheat farmer was a builder of categories; to him only wheat and oxen were useful. ...he failed to see the downward wash of over-wheated loam. When soil wash... finally put an end to wheat farming, Y and his like had already travelled far down the watershed'.
What does eutrophication cost? The answer must be an enormous sum in supplementary fertilizer for the land, treatment costs of waste water and domestic water supplies, losses of value of recreational and amenity waters. Usually the costs are internalized and passed on, hidden, to consumers. One way of looking at the economics of eutrophication and other environmental damage is in terms of benefits that undisturbed systems provide as opposed to the costs that accrue from disturbance of them. This gives an estimate of the value of such systems and of the cost to us of damaging them so that they no longer provide their present functions. Costanza et al (1997) have valued the services provided by natural ecosystems on a world scale at between US$ 16-54 trillion with a mean estimate of US$ 33 trillion which is 1.8 times the current global GNP (gross national product). In this analysis, services provided in nutrient cycling, acquisition and storage figure most prominently among 17 categories of goods and services. Services provided in waste treatment are the third most important after cultural services. Wetlands are particularly valuable in nutrient retention and are valued annually at nearly 15000 US$ per hectare, with freshwater swamps and floodplains and estuaries accorded the greatest values of all sixteen major habitats valued. Lakes and rivers are valued at $8500 per ha per year and freshwater habitats as a whole contribute about a fifth of the global total value of annual services provided.

Many of these habitats may be impaired in function by eutrophication - the loss of extensive areas of emergent reeds or submerged plants in shallow wetland lakes, for example. The cost of this loss is rarely counted, nor is that of possible health problems arising from high nitrate concentrations. There are encouraging signs however of communities realising the costs. The extensive cyanophyte blooms experienced in recent years in Australia (Bowling & Baker 1996) have led to much investment in phosphate precipitation at waste water treatment works. To minimise the costs to the community of chemicals for stripping and to minimise the transfer of nutrients to waterways through other routes, many local New South Wales communities have adopted a six-point domestic plan of action: (i) wash vehicles only on porous surfaces, (ii) fertilize lawns and gardens sparingly, (iii) compost food and garden waste,(iv) use zero or low phosphorus detergents, (v) do not use washing machines until a full load is ready, (vii) collect and bury pet faeces. Farmers are also encouraged to minimise fertilisation and cultivation. The efficacy of these measures has yet to be shown but there is no doubt that they represent an increasing awareness of the problem and resolve to solve it, not least because of the costs involved.

EVENTUALITIES AND ENVIRONMENT

Much of lowland north western Europe is a highly eutrophicated landscape as well as waterscape. The levels of fertilisation on arable fields and especially temporary grassland grown for silage are very large indeed. For many years we have been aware that much nitrate ran off cultivated and fertilised land, despite attempts to minimise it by better practice. Phosphorus, however, was presumed to be held more tightly in soils. There is increasing evidence, however, that soils have often now become saturated and phosphate is being discharged from them to replace, in waters, that which is now prevented from entering by greater attention to phosphate removal at waste water treatment works Foy et al 1995, Haygarth & Jarvis 1997, Heckrath et al 1995). In the UK it is clear that nutrient loads in small rural catchments have about doubled (Johnes et al 1997) since the pre-second world war period due to increased fertilisation levels and enhanced numbers of stock, practices encouraged by the subsidy system of the Common Agricultural Policy. There is comfort perhaps in the fact that most lowland waters are now so heavily eutrophicated that the situation is unlikely to worsen greatly but can only improve as nutrient control policies become more acceptable and take effect. The problem now transfers itself to one of how the extremes of eutrophication may be avoided in the newly intensifying areas of Eastern Europe, where many diverse habitats still exist, and the tropical world. Ljklema(1995b) has pointed out the startling difference in nutrient budget between a Dutch dairy farm and an arable holding in Rwanda.

There was a large surplus stored in the soil, lost by denitrification or leached in the Dutch case and an equally dramatic depletion, even without allowance for erosion and leaching in the Rwandan case. There appears to be a widespread deficiency of nutrients on many tropical farms which will generate pressure for heavy fertilisation in the future to increase production as populations continue to rise. Lijklema has expressed concern that this will lead to a eutrophication problem in tropical waters that are yet unaffected. This concern is well placed for there appears already to be a developing problem, expressed not least in large growths of water hyacinth, in Lake Victoria, which is among the more prominent of African lakes. The concern should perhaps be even greater, if Flannery (1994) is correct in his hypothesis that the use of European methods of agriculture is inappropriate on areas of ancient well-weathered soils.

He suggests that European soils are essentially very young, naturally nutrient-rich soils following the effects of glaciation in generating an abundance of weatherable minerals. North temperate climates also show only small seasonal precipitation extremes and support crop plants that are essentially 'competitors', able to sequester large amounts of nutrients from rich soils. In areas that have not been recently disturbed by glaciation, such as most parts of Africa, Australia and tropical South America, soils have been leached for such long periods that they are unable to sustain intensive agriculture, even with heavy fertilizer inputs because of their low organic content and propensity to erode in climates with strongly contrasting wet and dry seasons. Plants well fitted to such regimes tend to be unproductive specialists which compete poorly but survive extremes. Such strategies are inappropriate for agriculture.

Lijklema (1995b) is concerned that export of tropical products to the western world increases the nutrient depletion of tropical soils through nutrient export and is concerned that the inevitable consequence will be heavy investment of fertiliser in the tropics and severe eutrophication if all possible means of preventing it are not used. Flannery's (1994) position is that such agricultutre is unsustainable and should not be pursued in tropical soils. The implication is that most of world food production must shift to the areas where it can be sustainably maintained. This might mean even greater nutrient losses to north-temperate watercourses if the full barrage of techniques for preventing and minimising it is not brought to bear. These include suitable cultivation practices, buffer zones, and perhaps development of attitudes that discourage the enormous waste of food presently manifest in the western world through an emphasis on processed food and meat products, sometimes themselves from animals fed on high protein waste from other animals. It ought to be possible to feed a larger number of people from less intensively cultivated fields if culinary and dietary habits can be influenced towards less factory processing and a greater proportion of plant products. This is perhaps another example of the emerging view that the solution of environmental problems, such as eutrophication will not be possible without major attention to their ultimate causes in the way that societies are organised and manipulated by their most powerful members. Whether Chief Seattle really did say it or not in 1885, all things are connected. Aldo Leopold knew it too.

EDUCATION AND ENDING

Leopold ends his essay when atom Y has become rapidly trapped in sediment, with what sounds like complacency: '*Roots still nose among the rocks. Rains still pelt the fields....Black and white buffalo pass in and out of red barns, offering free rides to itinerant atoms*'. More likely it was acceptance that events unfold slowly and might be influenced by education and example. Leopold practised the land ethics that he preached. He was ahead of his time in realising just how much land has been abused with losses in fertility on the one hand and the consequences of eutrophication on the other. He managed his own farm in Sand County, Wisconsin, according to his principles whilst teaching at the University of Wisconin. Therein lies the key to bringing about desirable environmental change. There must first be personal conviction and a willingness to challenge the *status quo*, a passion to see a solution to the problems, and a willingness to transmit the information and the experience. These are the necessary features of University professors - not an ability to raise grant and contract money or manipulate committees. They were the characteristics of Aldo Leopold; they are why we hold this symposium to mark the retirement of Professor Lijklema.

REFERENCES

Barnden,A.D. (1993) Controlling nitrogen in the environment-feasibility and implications. National Rivers Authority, anglian region, Peterborough,UK.

Beklioglu,M. & Moss,B. (1996a) Existence of a macrophyte-dominated clear water state over a very wide range of nutrient concentrations in a small, shallow lake. Hydrobiologia 337, 93-106.

Beklioglu,M. & Moss,B. (1996b) Mesocosm experiments on the interaction of sediment influence, fish predation and aquatic plants with the structure of phytoplankton and zooplankton communities. Freshwater Biology 36, 315-325.

Bowling,L.C. & Baker, P.D. (1996) Major cyanobacterial bloom in the Barwon-Darling river, Australia in 1991 and underlying limnological conditions. Austraian J Mar. freshwat Res. 47, 643-657.

Costanza,R., d"Arge,R., de Groots,R., Farber,S., Grasso,M., Hannon,B., Limburg,K., Naeem,S., O'Neill,R.V., Paruelo,J., Raskin,R.G., Sutton,P. & van den Belt,M. (1997) The value of the world's ecosystem services and natural capital. Nature 387, 253-260.

Flannery,T.(1994) The Future Eater. Reed Books, Chatswood, New South Wales, Australia.

Foy,R.H., Smith,R.V., Jordan,C. & Lennox,S.D. (1995) Upward trend in soluble phosphorus loadings to Lough Neagh despite phosphorus reduction at sewage treatment works. Water research 29, 1051-1063.

Gibbons,B. (1981) Aldo Leopold. A durable scale of values. National Geographic 160(5) 682-708.

Hanssen,M.& Marsden,J. (1984) E for additives. Thorsons Publishers, Wellingborough

Haygarth, P.M. & Jarvis,S.C. (1997)Soil-derived phosphorus in surface runoff from grazed grasslandlysimeters. Water research 31,140-148.

Heckrath,G., Brookes,P.C., Poulton,P.R. & Goulding,K.W.T. (1995) Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. Journal of Environmental Quality 24, 904-910.

Johnes,P.J. (1996) Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. Journal of Hydrology 183, 323-349.

Johnes,P., Moss,B. & Phillips,G. (1996) The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: testing of a model for use in conservation and water quality management. Freshwater Biology 36, 451-473.

Jones,R.I., Young,J.M., Hartley,A.M. & Bailey-watts,A.E. (1996) Light limitation of phytoplankton development in an oligotrophic lake - Loch Ness, Scotland. Freshwater Biology 35, 533-543.

Leopold,A. (1949) A Sand County Almanac. Oxford University Press.

Lijklema,L. (19930 Considerations in modelling the sediment-water exchange of phosphorus. Hydrobiologia 253, 219-231.

Lijklema,L. (1994) Nutrient dynamics in shallow lakes: effects of changes in loading and role of sediemnt-water interactions. Hydrobiologia 275/276, 335-348.

Lijklema,L. (1995a) Water quality standards: sense and nonsense. Water Science & Technology 31, 321-327.

Lijklema, L. (1995b) Development and eutrophication: experiences and perspectives. Water Sci & Technol. 31, 11-15.

Mc Gowan, S. (1997) *Ancient cyanophyte blooms - studies on the paleolimnology of White Mere and Colemere*. PhD Thesis, University of Liverpool.

Moss,B. (1969) Limitation of algal growth in some central African waters. Limnol. Oceanogr. 14, 591-601.

Moss,B., Bekioglu,M., Carvalho,L., Kilinc,S., McGowan,S. &Stephen,D. (1997) Vertically challenged limnology; contrasts between deep and shallow lakes. Hydrobiologia 342/343, 257-267.

Moss,B., Johnes,P.J. & Phillips,G.L. (1996) The monitoring of ecological quality and the classification of standing waters in temperate regions - a review and proposal based on a worked scheme for British waters. Biological Reviews 71, 301-339.

Moss,B., Johnes,P.J. & Phillips,G.L. (1997) New approaches to monitoring and classifying standing waters.118-133 in P.J. Boon & D.L. Howell, eds, Freshwater Quality:Defining the Indefinable. Scottish National Heritage, Edinburgh

Moss,B., McGowan,S. & Carvalho,L. (1994) Determination of phytoplankton crops by top-down and bottom up mechanisms in a group of English lakes, the West Midland meres. Limnol. Oceanogr. 39, 1020-1029.

National Academy of Sciences (1969) Eutrophication: Causes, Consequences,Correctives. National Academy of Sciences, Washington,D.C.

Ozimek,T., Gulati,R.D., Van Donk,E. (1990) Can macrophytes be useful in biomanipulation of lakes. The lake Zwemlust example. Hydrobiologia 200/201, 399-407.

Petersen,R.C. (1992) The RCE, a riparian, channel and environmental invntory for small streams in the agricultural landscape. Freshwater Biology27, 295-306.

Reynolds C.S., 1979. The limnology of the eutrophic meres of the Shropshire -Cheshire plain - a review. Field Stud., 5: 93-173.

Sas,H. (ed) (1989) Lake restoration by reduction of nutrient loadings: expectations, experiences, extrapolations. Academia Verlag Richarz, Saint Augustin.

Schindler,D.W. (1977) The evolution of phosphorus limitation in lakes. Science 195, 260-262.

Talling,J.F. & Talling, I.B. (1965) The chemical composition of African lake water. Int.Rev. ges Hydrobiol. 50, 421-463.

Wright,J.F. (1995) Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. Austr.J.Ecol., 20, 181-197.

TOXIC ANOREXIA OF DAPHNIDS: A CRITICAL FACTOR IN THE DEVELOPMENT OF EUTROPHICATION PROBLEMS

Martin C.Th. Scholten, Edwin M. Foekema, Robbert G. Jak & Henno P. van Dokkum

TNO Environmental Sciences, Energy Research & Process Innovation, Dept. for Ecological Risk Studies, P.O. Box 57, 1785 AB, Den Helder, The Netherlands. tel. +31 223 638800; fax +31 223 630687; email LAMR@mep.tno.nl

ABSTRACT

Eutrophication problems in surface waters are generally considered to be the result of progressive fertilisation, although they are actually the result of an imbalance between (progressive) algal production and algal consumption by zooplankton. Semi-field multispecies ecotoxicity studies at mesocosm scale reveal that grazing of phytoplankton by daphnids is often the most sensitive species interaction in an aquatic community. Almost all toxic substances (except herbicides) first reduce the grazing efficiency of daphnids: a phenomenon which we have named "Toxic anorexia". A multispecies indoor microcosm assay has been developed to test this effect, under controlled conditions. In tests with water sampled from the field, toxic anorexia has been demonstrated comparing different lakes. Based on these observations, a conceptual model is presented to describe the interaction of nutrients and toxicants in the development of algal blooming. Eutrophication problems have to be considered as a result of progressive fertilisation in combination with sub-optimal daphnid functioning. The consequence is that reduction of the environmental nutrient load is not always the most effective method of precluding algal blooms.

KEYWORDS

Algae, eutrophication, grazing, management, toxicants, toxic anorexia, zooplankton

INTRODUCTION

Eutrophication problems in surface waters are generally considered to be the result of progressive fertilisation (e.g., Rijsdijk, 1996), although they are actually the result of an imbalance between (progressive) algal production and algal consumption by zooplankton (Figure 1).

National Academy of Sciences (1969) Eutrophication: Causes, Consequences,Correctives. National Academy of Sciences, Washington,D.C.

Ozimek,T., Gulati,R.D., Van Donk,E. (1990) Can macrophytes be useful in biomanipulation of lakes. The lake Zwemlust example. Hydrobiologia 200/201, 399-407.

Petersen,R.C. (1992) The RCE, a riparian, channel and environmental invntory for small streams in the agricultural landscape. Freshwater Biology27, 295-306.

Reynolds C.S., 1979. The limnology of the eutrophic meres of the Shropshire -Cheshire plain - a review. Field Stud., 5: 93-173.

Sas,H. (ed) (1989) Lake restoration by reduction of nutrient loadings: expectations, experiences, extrapolations. Academia Verlag Richarz, Saint Augustin.

Schindler,D.W. (1977) The evolution of phosphorus limitation in lakes. Science 195, 260-262.

Talling,J.F. & Talling, I.B. (1965) The chemical composition of African lake water. Int.Rev. ges Hydrobiol. 50, 421-463.

Wright,J.F. (1995) Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. Austr.J.Ecol., 20, 181-197.

TOXIC ANOREXIA OF DAPHNIDS: A CRITICAL FACTOR IN THE DEVELOPMENT OF EUTROPHICATION PROBLEMS

Martin C.Th. Scholten, Edwin M. Foekema, Robbert G. Jak & Henno P. van Dokkum

TNO Environmental Sciences, Energy Research & Process Innovation, Dept. for Ecological Risk Studies, P.O. Box 57, 1785 AB, Den Helder, The Netherlands. tel. +31 223 638800; fax +31 223 630687; email LAMR@mep.tno.nl

ABSTRACT

Eutrophication problems in surface waters are generally considered to be the result of progressive fertilisation, although they are actually the result of an imbalance between (progressive) algal production and algal consumption by zooplankton. Semi-field multispecies ecotoxicity studies at mesocosm scale reveal that grazing of phytoplankton by daphnids is often the most sensitive species interaction in an aquatic community. Almost all toxic substances (except herbicides) first reduce the grazing efficiency of daphnids: a phenomenon which we have named "Toxic anorexia". A multispecies indoor microcosm assay has been developed to test this effect, under controlled conditions. In tests with water sampled from the field, toxic anorexia has been demonstrated comparing different lakes. Based on these observations, a conceptual model is presented to describe the interaction of nutrients and toxicants in the development of algal blooming. Eutrophication problems have to be considered as a result of progressive fertilisation in combination with sub-optimal daphnid functioning. The consequence is that reduction of the environmental nutrient load is not always the most effective method of precluding algal blooms.

KEYWORDS

Algae, eutrophication, grazing, management, toxicants, toxic anorexia, zooplankton

INTRODUCTION

Eutrophication problems in surface waters are generally considered to be the result of progressive fertilisation (e.g., Rijsdijk, 1996), although they are actually the result of an imbalance between (progressive) algal production and algal consumption by zooplankton (Figure 1).

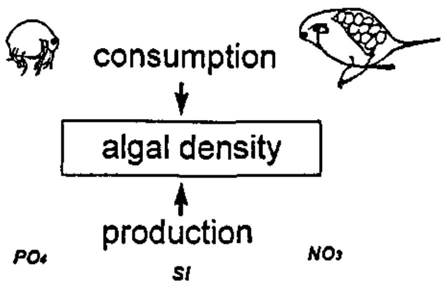


Figure 1. Algal density as resultant of production and consumption.

Semi-field multispecies studies at mesocosm scale reveal that grazing of phytoplankton by daphnids is often the most sensitive species interaction in an aquatic community (deNoyelles *et al.*, 1994). Toxic effects upon zooplankton species in toxicity tests are generally evaluated on the basis of short-term toxicity tests, in which the mortality is caused by high exposure concentrations, or in long term (semi-) chronic tests, in which effects upon the reproductive output are considered. The latter tests involve lower exposure concentrations but are very elaborate and expensive. Sublethal effects due to very low exposure concentrations may be caused by reducing the feeding efficiency or toxic anorexia, as we name it (Figure 2).

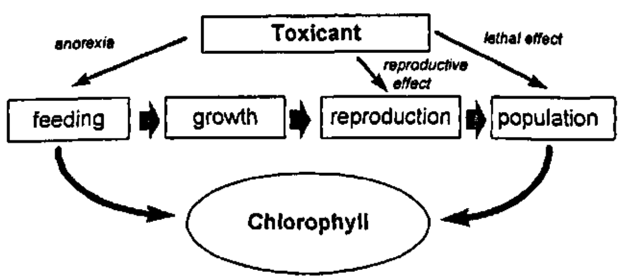


Figure 2. Conceptual model of effects of toxicants on daphnids, and consequences for algal density

By lowering the uptake of algae by daphnids, exposure to toxicants thus result in an imbalance between algal production and algal consumption, leading to the development of enhanced algal densities and subsequent eutrophication problems (toxic or nuisance algae; turbid waters; anoxic conditions; bad smell, midge plagues etc.) under conditions of prolific algal production. This paper summarizes research performed over the past few years at the TNO Laboratory for Applied Marine Research, which demonstrates toxic anorexia. The paper concludes with a conceptual model outlining the interaction between nutrients and toxicants in relation to eutrophication, with recommendations for water management.

MESOCOSM OBSERVATIONS: ALGAL BLOOMING DUE TO TOXICANTS.

In experiments with enclosed plankton communities (Yasuno *et al.*, 1993; Jak *et al.*, 1996; Jak *et al.*, 1997) it was observed that the development of the algal density is correlated with the toxic inhibition of daphnid population development (Figure 3). Daphnid species with the lowest algal threshold level showed the strongest reduction by toxicants (*Ceriodaphnia* < *Daphnia* < *Chydorus*). Less efficient grazers (viz. copepods and rotifers) benefit from reduced competition. The experiments showed that a cyanobacteria bloom could be induced by the presence of toxicants, while under pristine (but eutrophic) conditions daphnids could control the algal density at low levels.

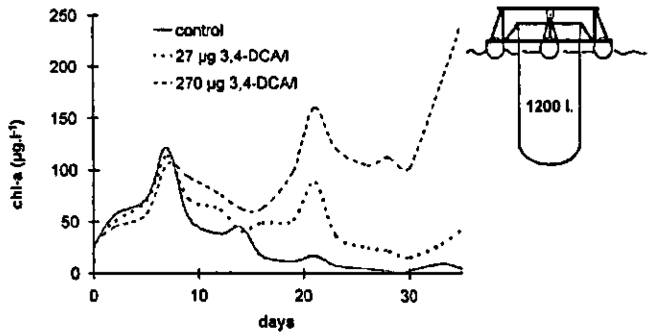


Figure 3. Effects of two concentrations of 3,4 DCA on algal density (expressed as concentration chlorophyll-a, in µg.l^{-1}) in an experiment in 1200 l enclosures, compared to a control (Redrawn from *Jak et al.*, 1997)

In long term experimental pond studies on the ecological effects of polluted sediments (TNO, unpublished data), the most sensitive response to moderately polluted sediments appeared to be the development of the algal density during spring, finally resulting in a summer bloom of cyanobacteria (Figure 4). The nutrient loads of the two systems were comparable, but the Ketelmeer sediment contained considerably higher pollutant levels. The algal density corresponded with the final cladoceran: copepod ratio in the early summer; high algal densities resulted from poor cladoceran functioning.

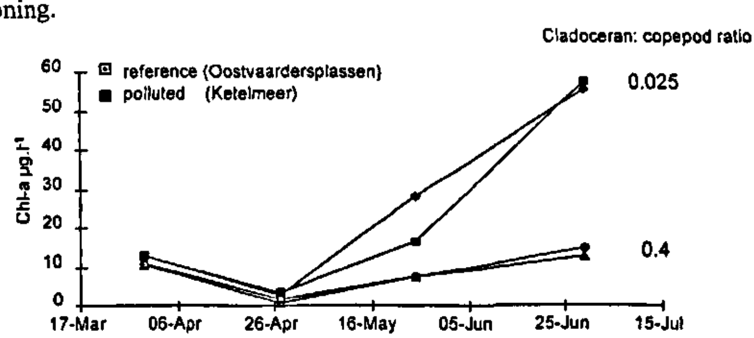


Figure 4. Phytoplankton development (expressed as concentration chlorophyll-a, in µg.l^{-1}) in experimental ponds with polluted sediment (from lake Ketelmeer) and reference sediment (Oostvaardersplassen) (TNO, unpublished data)

A mesocosm test with the organophosphorus insecticide dimethoate (Figure 5, Foekema *et al.*, 1996) also indicated reduced grazing of phytoplankton by daphnids as the most sensitive species interaction, resulting in stimulated growth of filter-feeding benthic bivalves such as *Sphaerium corneum*. Under the oligotrophic test condition (5 μg P.l^{-1}), this did not result in prolific algal densities. However, in additional tests with plankton samples in indoor microcosms under (fertilised) mesotrophic conditions (50 μg P.l^{-1}) it was demonstrated that the induction of algal blooming could be related to the toxic inhibition of the daphnids.

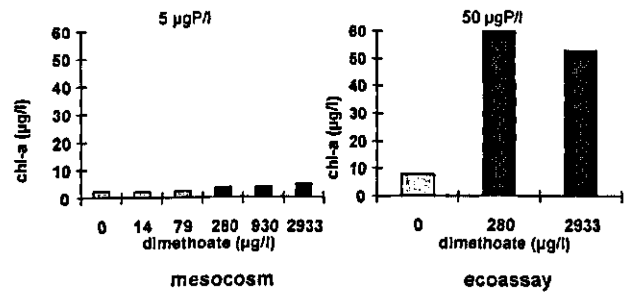


Figure 5. Effect of dimethoate on algal density (expressed as concentration chlorophyll-a, in μg.l^{-1}), at two fertilisation levels (from Foekema *et al.*, 1996)

PLANKTON ECOASSAY: TESTING TOXIC ANOREXIA OF DAPHNIDS

In order to test the toxic anorexia of daphnids, a "plankton ecoassay" test procedure has been developed. Toxic anorexia of daphnids is established in 80 liter test containers installed in a temperature controlled room. In a hypertrophic medium stocked with cultured algae (*Chlorella*, *Scenedesmus* or others), prolific algal production rates are realised. An assemblage of daphnid-species (i.e. *Daphnia*, *Ceriodaphnia*, *Simocephalus*) at a density of 5-10 per litre are tested in these systems for their capacity to consume the algae. The balance between algal production and consumption is reflected by the development of the algal density. Under pristine reference conditions, the daphnids are capable of controlling algal densities at a level of 5-10 μg.l^{-1} chlorophyll. An increase of the algal density indicates toxic inhibition of grazing by daphnids (toxic anorexia).

An example experimental outcome is presented in Figure 6 (for dimethoate, TNO, unpublished data). The algal density development is exponential in absence of daphnids, while daphnids can control the density at a low level. Even at 20μg.l^{-1} dimethoate, this grazing control fails. In comparison: The NOEC for a 21 day Daphnid growth or reproduction test is ca. 30μg.l^{-1} dimethoate.

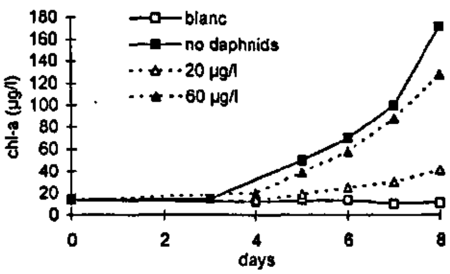


Figure 6. Development of phytoplankton in microcosms with two different levels of dimethoate, compared to a pristine reference microcosm and a microcosm without zooplankton (TNO, Unpublished data).

The plankton ecoassay enables the establishment of a NOEC or EC_{50} for substances or effluent dilution series for one of the most critical processes (species interaction) in an aquatic ecosystem, representing the toxicity at the community level.

The plankton ecoassay has been applied in the testing of toxic anorexia in field samples of surface waters (including natural phytoplankton, daphnids excluded). The example given below shows that daphnids from a laboratory culture added to the field samples are not capable of controlling algal densities in "Lake Amstelmeer", as daphnids do in the "Lake Geestmerambacht". This indicates a (as yet unresolved) water quality problem in the Amstelmeer.

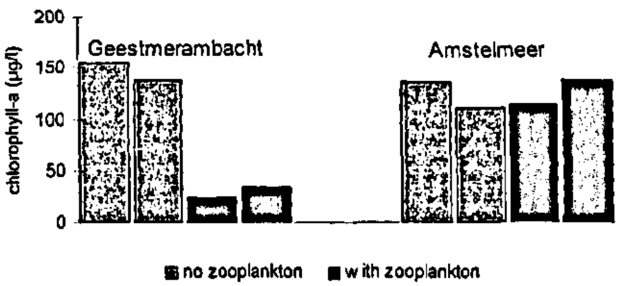


Figure 7. Algal densities in microcosms containing water from (the relatively pristine) lake Geestmerambacht and lake Amstelmeer (relatively polluted), with and without Daphnids (TNO, unpublished data)

The cladoceran: copepod ratio (observed in field samples) is 0.15 in the Geestmerambacht and 0.05 in the Amstelmeer. In indoor microcosms, lacking fish predation, this ratio develops within a week up to 2-5 for Geestmerambacht zooplankton, but remains 0.05 for Amstelmeer zooplankton.

A pilot plankton ecoassay with water samples taken from various polder waters showed an inverse relationship of daphnid grazing with sum toxic units of pesticides (Figure 8). The grazing efficiency was determined by removing the zooplankton from natural surface water, and comparing the algal density of microcosms without and with added daphnids. Although these results are preliminary, they indicate the occurrence of toxic anorexia in the field situation. In pristine reference water, grazing efficiencies of 80% have been observed.

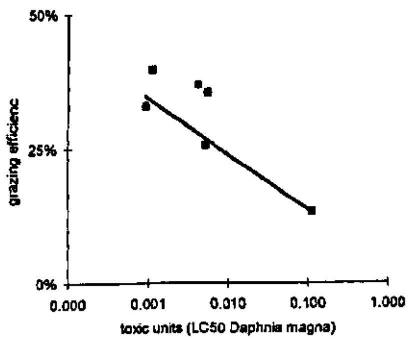


Figure 8. Grazing efficiency of Daphnids in relation to the contamination of suface waters (polder waters).

TOXICANTS AND EUTROPHICATION: CONCEPTUAL MODEL

The plankton ecoassay enables us to test the response of the plankton community to various levels of nutrition. Fertilisation tests showed that the relation between phosphorus (and other nutrients) and chlorophyll strongly depends on the grazing efficiency of daphnids. A plankton ecoassay test of a fertilisation series under presence or absence of daphnids (TNO, unpublished data) shows a good consistency with field observations reported by Sarnelle (1992) (Figure 9).

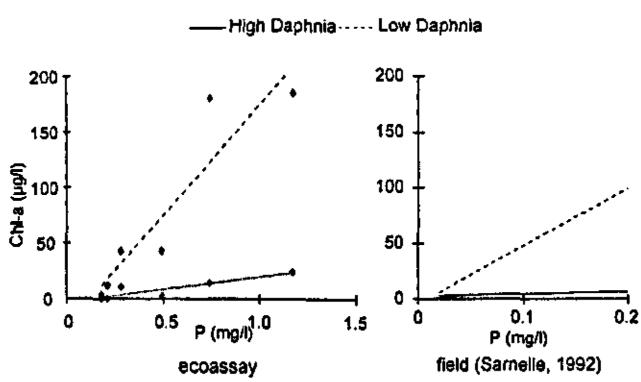


Figure 9. Development of phytoplankton under high and low zooplankton densities, in microcosms (left) and a field situation.

Based on these observations a conceptual model is constructed to describe the interaction of nutrients and toxicants with regards to the development of algal blooming (Figure 10). Nutrients will lead to an increased algal production. In pristine conditions, this will not lead to a large increase in algal blooming potential, because the algal consumption by daphnids can counter the increased production up to a certain limit. However, in polluted situations, the algal blooming potential increases rapidly with increasing algal production, due to the absence of grazing control.

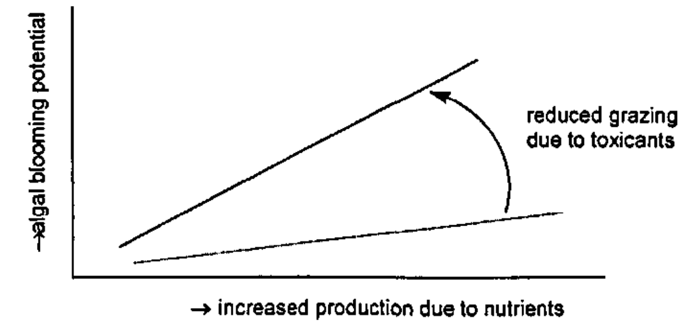


Figure 10. Interacting effects of nutrients and toxicants on the development of algal blooms

This conceptual model is supported by fertilisation tests with field samples from lake Geestmerambacht and lake Amstelmeer (TNO, unpublished data, see Figure 11).

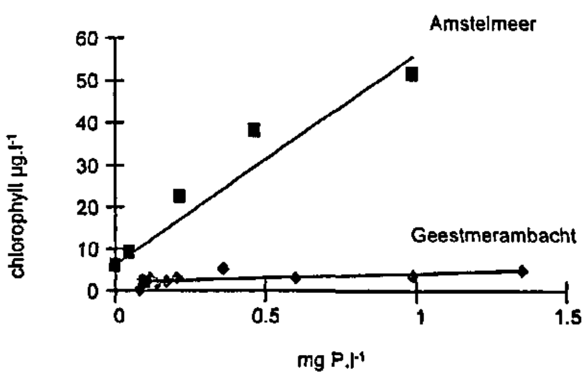


Figure 11. Algal density at increasing fertilisation levels in lake Geestmerambacht and lake Amstelmeer.

CONCEQUENCES FOR WATER MANAGEMENT

The importance of algal grazing in aquatic ecosystems is based on the fact that in a healthy, productive ecosystems the algal productions is almost completely consumed by herbivores. The production of grazers is approximately 5% of the primary production, in comparison to 1% in terrestrial ecosystems.

A small reduction in algal grazing efficiency due to sublethal toxicity (toxic anorexia) can result in an strong increase of the algal density under productive conditions. At high algal densities, nuisance colony forming and inedible algae become dominant, which results in a state which is experienced as eutrophication.

Eutrophication problems have to be considered as a result of progressive fertilisation in combination with sub-optimal daphnid functioning. Toxic anorexia is an important factor to be acknowledged. Depending on the local situation, either nutrient reduction or pollution reduction, or a combination of these might prove to be the most cost-effective measure in eutrophication problems. Therefore, it is recommended that before large scale (and expensive) eutrophication combatment programs are implemented, a good insight in the eutrophication status of an aquatic ecosystem should be obtained, not only in terms of nutrient load and algal density, but also in terms of production and grazing. The plankton ecoassay provides a useful tool for this.

REFERENCES

deNoyelles F., S.L. Dewey, D.G. Huggings & W.D. Kettle (1994): Aquatic mesocosms in ecological effects testing: Detecting direct and indirect effects of pesticides. In: R.L. Graney, J.H. Kennedy & J.H. Rodgers (eds.), Aquatic mesocosm studies in ecological risk assessment. SETAC Special Publication. Lewis Publishers, Boca Raton. pp. 577-603.

Foekema E.M., R.G. Jak & M.C.Th. Scholten (1996), Toetsing van enkele veldexperimentele technieken in microcosms behandeld met dimethoaat. TNO-MEP R96/070.

Jak R.G., J.L. Maas & M.C.Th. Scholten (1997), Ecotoxicity of 3,4-dichloroaniline in enclosed freshwater plankton communities at different nutrient levels. Ecotoxicology (in press).

Jak R.G., J.L. Maas J.L. & M.C.Th. Scholten (1996), Evaluation of laboratory derived toxic effect concentrations of a mixture of metals by testing fresh water plankton communities in enclosures. Water Res. 30:1215-1227.

Rijsdijk R.E. (1996): Mogelijke maatregelen bij de bestrijding van eutrofiëring in Nederlandse plassen en meren. RIZA Nota Nr. 96.049C. 95pp.

Sarnelle O. (1992): Nutrient enrichment and grazer effects on phytoplankton in lakes. Ecology 73:551-560.

Yasuno M., A. Asaka & Y. Kono (1993), Effects of pyraclofos (an organophosphorous insecticide) on nutrient enriched ecosystems. Chemosphere 27:1813-1824.

GROWTH INHIBITION OF *DAPHNIA* FEEDING ON NUTRIENT-LIMITED *SCENEDESMUS*

M. Lürling and E. Van Donk

Department of Water Quality Management & Aquatic Ecology
Agricultural University Wageningen
PO Box 8080, 6700 DD Wageningen, The Netherlands

ABSTRACT

The growth, reproduction and feeding of *Daphnia pulex* DeGeer on nutrient-limited *Scenedesmus acutus* Meyen was examined. In comparison with animals grown on nutrient sufficient cells, nutrient limited cells resulted in smaller body-length, reduced brood size, reduced age at maturity, increased age at first reproduction and consequently in reduced *Daphnia* population growth rates. Under nutrient-repletion *Scenedesmus* forms colonies when exposed to *Daphnia*, whereas under nutrient limitation the cells remain unicellular. These mechanisms affect the energy-transfer from algae to herbivorous zooplankton.

KEYWORDS

Daphnia, infochemicals, life-history, nutrient-limitation, *Scenedesmus*, trophic interaction

INTRODUCTION

Nutrient limitation is not a rare phenomenon restricted to oligotrophic waterbodies, but may even occur around algal blooms in eutrophic lakes. For example, both P-limitation (Riegman, 1985) and N-limitation (Sommer, 1989) have been observed in hypertrophic lakes. Also in the eutrophic lake Zwemlust (The Netherlands) with an average chlorophyll-a content of 26.7 (±23.3) $\mu g\ l^{-1}$ during 1996, very low nitrogen values were measured with an annual average (± 1SD) of 0.26 (±0.33) mg N l^{-1}, while the average phosphorus concentration was 0.69 (±0.20) mg o-P l^{-1}. Moreover, nutrient limitation may exhibit a high spatial and temporal variability (Butler *et al.*, 1989).

Variations in nutrient concentrations may influence the taxonomic and chemical composition of the phytoplankton. The quality of phytoplankton as food for zooplankton depends on morphological and biochemical features such as size (Burns, 1968) and toxicity (Lampert, 1987). An altered morphology and chemical composition may influence the ingestibility and digestibility of algal cells and, hence, may affect zooplankton growth (Van Donk *et al.*, 1997). Especially the non-selective grazers belonging to the genus *Daphnia* will be often confronted with the high variance in algal food quality. Nutrient-limited algae have been reported to reduce growth and reproduction in *Daphnia* (e.g. Groeger *et al.*, 1991; Mitchell *et al.*, 1992; Sterner *et al.*, 1993) and rotifers (Rothhaupt, 1995).

Many algae are highly plastic in morphology, growth and chemical composition and variable traits have been interpreted as defence mechanisms against grazing (Van Donk *et al.*, in press). The notoriously phenotypically plastic algal genus *Scenedesmus* (Trainor, 1991) is one of the commonest genera in freshwater habitats (Canter-Lund and Lund, 1995). Since herbivory is one of the main losses among algae (Sterner, 1989) a strong selection pressure exists on the development of traits that reduce mortality. In the presence of *Daphnia* the unicell-colony transformation in *Scenedesmus* is triggered in order to reduce their vulnerability against grazing (Hessen and Van Donk, 1993; Lampert *et al.*, 1994; Lürling and Van Donk, 1996). Various nutrient conditions, in the absence of *Daphnia*, did not affect colony size in *Scenedesmus* (Sterner and Smith, 1993).

In this study the changes in morphology and chemical composition of *Scenedesmus acutus* under N-and P-limitation and the effects on life-history parameters and the feeding behaviour of *Daphnia pulex* were investigated.

METHODS

Algae. The green alga *Scenedesmus acutus* Meyen (Chlorococcales; Chlorophyceae) was obtained from the Max-Planck Institute for Limnology (Plön, Germany). The algae were cultured in five separate 0.55 l chemostats with different dilution rates (Table 1) on 20% Z8 medium (Skulberg and Skulberg, 1990), and on 20% Z8 medium with modified N:P ratio's (Table 1).

Table 1.Nitrogen (μM) and phosphorus (μM) contents of nonlimited and nutrient limited Z8 (20%) medium, including the N:P ratio and chemostat dilution rate D (d^{-1}).

Limitation	N (μM)	P (μM)	N:P	D (d^{-1})
Nonlimited	1200	35	34	~1.0
N-moderate	400	35	11	~0.5
N-strong	120	35	3.4	~0.23
P-moderate	1200	15	80	~0.5
P-strong	1200	5	240	~0.23

The chemostats were continuously illuminated by circular fluorescent tubes (Philips TLEM 40W/33RS) at an irradiance of ~100 μmol quanta m^{-2} s^{-1} in a temperature controlled chamber at 20 ± 1°C. The algae were harvested daily from the overflow vessel and used as food for *Daphnia pulex*. Algal densities and size distributions were determined in the range 3.0 - 20.0 μm equivalent spherical diameter (ESD) using the Coulter Multisizer II (100 μm capillary). For each limitation the number of cells per colony was determined microscopically in subsamples preserved in Lugol's fixative. Cell dimensions (length and width) were measured using a Leica Quantimet 500 MC image-analyser coupled with a Nikon light-microscope at 600× magnification.

Chemical analysis. Algae were analysed for their C, N, P, total protein, carbohydrate and lipid contents. Algal C was determined using a modified chemical-oxygen-demand measurement (Golterman and Clymo, 1969). N- and P-contents were determined using a SKALAR autoanalyzer after sulphuric acid/peroxide digestion of the cells. Total protein content of the algae was determined according to the method described by Lowry *et al.*(1951). Carbohydrates were measured using the anthrone method (Hassid and Abraham, 1957). Total lipids were determined following the method of Meyer and Walther (1988).

Animals. The grazer *Daphnia pulex* originates from Lake Zwemlust (The Netherlands). A clonal population has been cultured in the laboratory for over a year at 20°C in 1 litre jars on a suspension of *S. acutus* in membrane filtered (through 0.45 μm) lake water.

Daphnia induced colony formation. The susceptibility of nonlimited, moderate N-limited and strong N-limited cells to colony inducing chemicals exuded by actively feeding *Daphnia* (see Hessen and Van Donk, 1993; Lampert *et al.*, 1994) was examined. Algae were harvested from the chemostats and 20 ml was inoculated into 30 ml of the corresponding medium. Three flasks served as controls, whereas three other flasks per algal limitation received 5 ml membrane filtered (0.1 μm) water from a *Daphnia pulex* culture (~200 animals l^{-1}). The experimental flasks were incubated for 48 h at 20°C on a rotating shaking table and continuously illuminated at ~100 μmol quanta m^{-2} s^{-1}. Initially and after 48 h cell densities and size distributions were determined in the range 3.0 - 20.0 μm ESD. Numbers of cells per colony were determined microscopically.

Life-history experiment. Prior to the experiment, one adult *Daphnia* was transferred into 0.45 μm filtered lake water with *S. acutus* as food. New-borns were collected within 24 h of birth and placed individually in 100 ml tubes. Animals were grown separately in the tubes by daily renewing the filtered lake water with algal food. New-borns from the third brood, born from different mothers within 20 h, were collected and joined in a 500 ml beaker. For each treatment 15 neonates were selected from this beaker and transferred individually into 100 ml test tubes containing 60 ml algal suspensions in filtered lake water. Because of dissimilar morphology and carbon content per algal cell, animals were fed with equivalent algal biovolumes (i.e. ~10^7 μm^3 ml^{-1}; ~ 3.5 mg C l^{-1}). The tubes were incubated in a temperature controlled room at 20°C in the dark. The animals were transferred daily into clean tubes containing fresh food suspensions. Conductivity and pH of the food suspensions were similar and on average (± 1 SD) 398 ± 23 μS cm^{-1} and 8.18 ± 0.12, respectively. Body-length (mm) of the animals was recorded by measuring them from above the eye to the base of the tail. Animals were examined daily for moulting. Time needed to reach maturity (d), the number of new-borns and survival (%) were also recorded. New-borns were removed from the tubes. The experiment was terminated after the animals had reached the fourth adult instar and consequently had released their third clutch. The intrinsic rate of population increase (*r*) was estimated using the Euler equation:

$$1 = \sum_{x=0}^{N} e^{-rx} l_x m_x \qquad (1)$$

where r = rate of population increase (d^{-1}), x = age class (0...N), l_x = probability of surviving to age x, and m_x = fecundity at age x. A Jack-knifing method was used to calculate standard errors of *r* (Meyer *et al.*, 1986). In order to compare values of *r* statistically, the data set of each treatment was split into two parts. Effects of nutrient limited algae on life history parameters were compared with one-way ANOVA ($\alpha = 0.05$).

Grazing experiments. Ten adult *Daphnia pulex* (belonging to the same cohort) were incubated for 3 h in 100 ml algal suspensions at 20°C in the dark. The bottles were incubated in triplicate and manually shaken every 30 min. The algal concentrations (i.e. biovolumes) were kept similar for each treatment and were on average 10^7 μm^3 ml^{-1}. Initially and after 3 h of grazing algal volumes were determined using the Coulter Multisizer II. Clearance rates (CR, ml $animal^{-1}h^{-1}$) were calculated according the relationship:

$$CR = \left(\frac{[\ln V_0 - \ln V_t]}{\Delta t}\right) \times \left(\frac{v_v}{N}\right) \qquad (2)$$

where V_0 = the initial algal volume (μm^3), V_t = the volume after 3 h, Δt = the feeding time (3 h), v_v = the experimental vessel volume (ml) and N is the number of animals per vessel. Clearance rates were statistically compared using one-way ANOVA, followed by a Tukey's test.

RESULTS

Algal food characteristics. The algal growth limitations are reflected in the atomic C:N and C:P ratios (Table 2). The C:N- or C:P-ratio was highest when algae were grown in medium with lowest N-content or P-content, respectively. Closely related to the elemental composition is the biochemical make-up of the algae. Nutrient-limitation resulted in an increase in carbohydrates and a decrease in proteins (Figure 1).

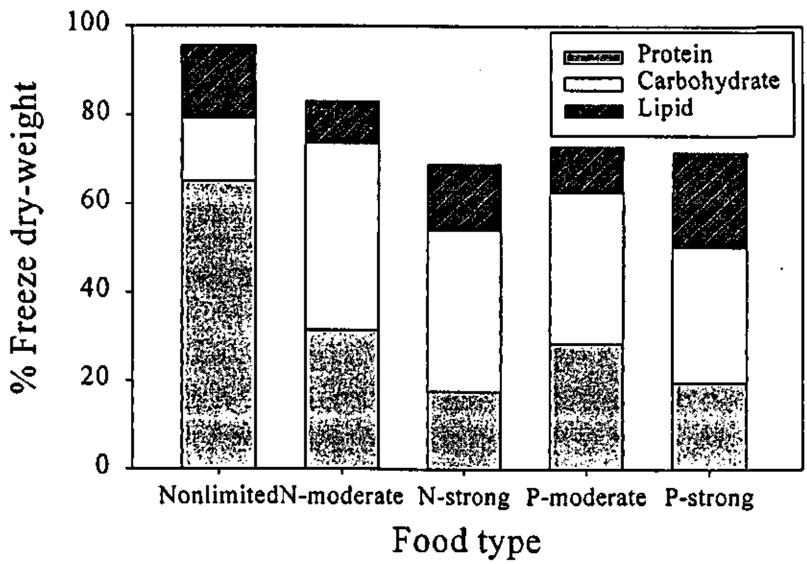


Figure 1. Biochemical composition (total protein, carbohydrate and lipid) of non-, N-, and P-limited *Scenedesmus* as a percentage of freeze dry-weight.

Also morphological features as cell dimensions and cell volume changed when *Scenedesmus* was cultured in N- or P-limited medium. However, no colony formation was observed (Table 2).

Table 2. Elemental ratios C:N and C:P on atomic basis (mol/mol), cell dimensions (length × width ± 1 SD; μm), mean particle volume (± 1 SD; μm^3) and mean number of cells per colony (± 1 SD).

Scenedesmus	C:N	C:P	Length × Width	Volume	Cells per colony
Nonlimited	4.5	112.3	13.6 (1.2) × 3.9 (0.7)	117.4 (22.3)	1.43 (0.03)
N-moderate	7.7	216.8	11.1 (1.6) × 4.3 (0.6)	117.9 (14.3)	1.92 (0.14)
N-strong	13.7	313.6	11.0 (2.7) × 5.5 (0.7)	183.7 (22.7)	1.10 (0.06)
P-moderate	4.1	303.2	8.9 (1.6) × 5.1 (0.8)	172.8 (21.5)	1.42 (0.17)
P-strong	4.9	903.5	9.3 (1.3) × 6.3 (0.8)	335.7 (44.2)	1.59 (0.08)

Daphnia induced colony formation. When exposed to 10% (v/v) filtered water from a *Daphnia* culture formation of colonies was induced only in nonlimited and moderately limited cells, strongly limited *Scenedesmus* showed no response (Table 3). Hence, colony formation seems to be restricted to actively growing cells.

Table 3. Effect of *Daphnia* water (DW) on colony size of nonlimited and nutrient-limited *Scenedesmus*. Asterisks (*) sharing the same vertical column indicate homogeneous groups that are not significantly different at the 95 % level (Tukey's test), including F and p-values of two-way ANOVA.

Scenedesmus acutus	Cells per colony				Two-way ANOVA		
Nonlimited	1.74 (0.03)	*				F	p
Nonlimited + DW	4.98 (0.15)			*	Algal type	38.2	<0.001
N-moderate	3.35 (0.35)		*		*Daphnia* water	90.1	<0.001
N-moderate + DW	5.28 (0.22)			*	Interaction	19.3	0.002
N-strong	2.10 (0.67)	*	*				
N-strong +DW	2.41 (0.17)	*	*				

Daphnia growth. Strong nutrient-limitation resulted in decreased body size in consecutive instars compared with animals grown on moderate limited or nonlimited *Scenedesmus* (Figure 2).

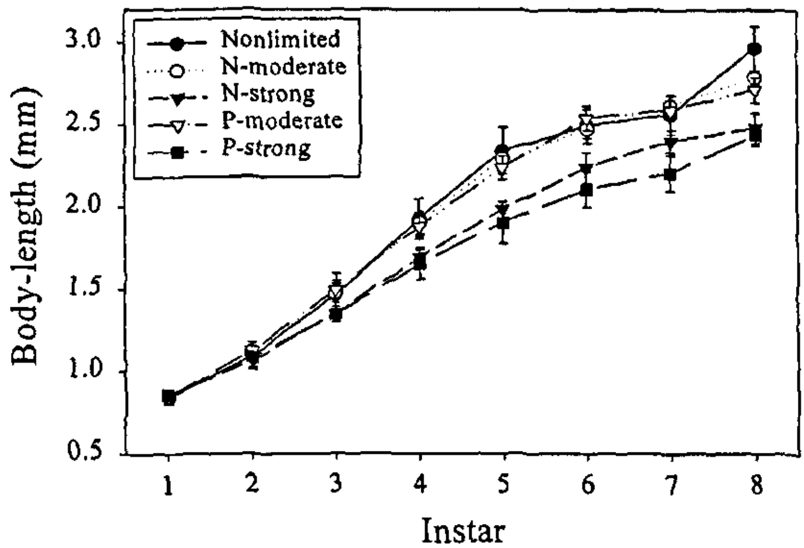


Figure 2. Body-length (mm ± 1 SD) of *Daphnia pulex* feeding on non-, N-, or P-limited *Scenedesmus acutus*.

Age and size at maturity were significantly affected (one-way ANOVAs: $F_{4,68}=9.20$; $p<0.001$ and $F_{4,68}=34.4$;$p<0.001$, respectively) by nutrient-limitation of the algal food (Table 4). Also the number of new-borns produced during the three observed adult instars was strongly influenced by nutrient-limited algae. Two-way ANOVA indicated a significant brood number ($F=14.2$; $p=0.002$) and food type effect ($F=8.3$; $p=0.006$). Reproductive rates were highest for animals feeding on nonlimited *Scenedesmus*, but lowest when strongly N- or P-limited were fed. (Figure 3). The effects of nutrient-limitation on somatic tissue growth, development and reproduction resulted in lower rates of population increase (*r*) when animals were fed nutrient-limited *Scenedesmus* (Table 4).

Table 4. Life-history parameters of *Daphnia pulex* feeding on nonlimited or nutrient-limited *Scenedesmus acutus*. Rates of population increase (r, d^{-1} ± 1SE of the Jack-knifing method), age at maturity (d) and size at maturity (mm). Asterisks (*) as in table 3.

Limitation	Age at maturity (d)				Size at maturity (mm)				Growth rate (d^{-1})			
Nonlimited	5.1 (0.5)	*			2.36 (0.10)	*			0.416 (0.006)	*	*	
N-moderate	5.4 (0.5)	*	*		2.28 (0.05)	*	*		0.450 (0.004)	*		
N-strong	6.4 (0.9)			*	2.09 (0.14)			*	0.362 (0.013)		*	*
P-moderate	6.0 (0.1)		*		2.24 (0.07)		*		0.448 (0.003)	*		
P-strong	6.5 (1.3)			*	2.00 (0.10)			*	0.274 (0.014)			*

Daphnia fed nonlimited algae became gradually orange-brownishly coloured, while animals fed strongly nutrient-limited algae remained pale. Also frequently an old moult was attached to the tail of limited animals, especially when strong P-limited cells were the food. Besides a significant effect of instar, i.e. age of the animals, ($F=43.0$; $p<0.001$), also nutrient limitation had a significant effect ($F=50.4$; $p<0.001$) on the length of this tail spine.

Grazing experiments. A short-term grazing experiment revealed significant differences ($F_{2,9}$=37.4; p<0.001) in clearance rates between animals that had been fed with nonlimited and strong nutrient-limited *Scenedesmus* (Table 5). A similar observation was made when *Daphnia* was fed with *S. acutus* of different age. The highest clearance rate were found on fresh, actively growing algae, intermediate on late-log phase algae (eight days old) and the lowest on stationary phase algae (23 days old).

Table 5. Clearance rates (± 1SD; in ml animal^{-1}h^{-1}) of *Daphnia* feeding on nonlimited, nutrient-limited and senescent *Scenedesmus* cells, including colony size of cells of different age. Asterisks as in table 3.

Food type	Clearance rate				Food type	Clearance rate			Cells per colony
Nonlimited	1.52 (0.22)	*			Nonlimited	1.19 (0.16)	*		1.23 (0.12)
N-strong	0.66 (0.16)		*		8 days	0.85 (0.22)	*		2.15 (0.14)
P-strong	0.25 (0.17)			*	23 days	0.42 (0.19)		*	1.91 (0.11)

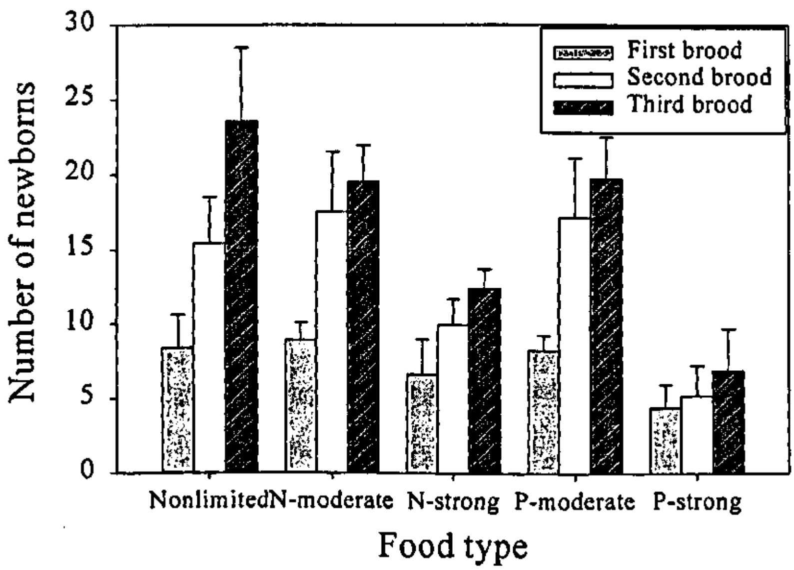


Figure 3. Number of new-born *Daphnia* in the first three broods by animals feeding on non-, N-, or P-limited *Scenedesmus acutus.*

DISCUSSION

Daphnia's growth and reproduction were strongly affected by the nutrient-limited *Scenedesmus* cells, resulting in decreased population growth rates. These results are comparable to growth of *Daphnia obtusa* on nutrient-limited *Scenedesmus* (Sterner *et al.*, 1993; Sterner, 1993). The observed effects such as reduced body-length, reduced brood size, and increased age at maturity resemble those as found under resource depression (Lynch, 1989).

A possible explanation for lower *r*-values may be a reduced feeding activity of the animals. The clearance rates of animals feeding on either senescent cells or nutrient-limited cells were remarkably reduced compared to feeding on nonlimited cells. Several studies have reported similar results of reduced clearance rates of *Daphnia* when feeding on senescent cells (Ryther, 1954) or nutrient-limited algae (Sterner and Smith, 1993; Sterner, 1993; Van Donk and Hessen, 1993). The reduced clearance rates may be the result of either a reduced intake of food, a reduced digestion of the cells, or a combination of both. However, a reduced food intake seems unlikely because *Daphnia* is believed to lack any ability to select food based on taste (DeMott, 1986). Therefore, a reduced digestibility is a possible cause. Lower digestibility rates may be the result of either an animals' lower gut activity or may be caused by an increased algal resistance against digestion. Mucous excretion or cell wall thickening may reduce the digestibility of algal cells and allow viable gut passage (Porter, 1975; Van Donk and Hessen, 1993; Van Donk *et al.*, 1997).

Another explanation for the lower *Daphnia* growth may be an altered chemical composition of the algae under nutrient limitation. The daphnids may suffer from a direct mineral limitation (Hessen, 1992; Sterner, 1993) or from a changed biochemical composition of the algae as a result of the nutrient limitation (Ahlgren *et al.*, 1990; Giani, 1991; Müller-Navarra, 1995). Sterner (1993) found that *Scenedesmus* high in carbohydrates were of low quality and cells high in protein content of good quality to *Daphnia*, unless P was low. Urabe and Wanatabe (1992) estimated a threshold food C:P ratio of 200-400 above which *Daphnia* growth will be reduced. Although the correlation between C:P and *r* was significant in our experiments ($r = 0.876$; $n = 5$; $p = 0.026$), this was mainly due to the strong-P-limited data point with a high leverage. Daphnids fed with strong-N-limited algae had, for example, a significantly lower *r* than animals fed with moderate P-limited algae while the C:P ratios of both algal populations were comparable, i.e. 314 and 303, respectively. The low protein content seems the most probable explanation for the observed difference. Thus, the C:P ratio may not be the sole predictor of *Daphnia* growth. Moreover, an altered internal chemistry of the algae can not explain the lower clearance rates of daphnids on nutrient-limited algae, unless some structural features as cell wall thickness are influenced (Van Donk *et al.*, 1997).

The mechanism of an increased grazing resistance of nutrient-limited cells has been interpreted as a defence mechanism governed by the algae (Mitchell *et al.*, 1992; Van Donk and Hessen, 1993). However, it is hard to believe that algal cells under nutrient depletion will be concerned about the presence of grazers. The algae simply respond to the nutrient limitation and a formation of a thicker cell wall has to be seen as a response to that rather than an inducible defence mechanism. It would be more obvious if algae would invest in a defence mechanism under nutrient replete conditions. The availability of sufficient nutrients could allow the formation of thicker cell walls. However, this seems not to be case. Pelagic algae may use two strategies to reduce grazing losses, either avoid being ingested or, if ingested, avoid being digested (Van Donk *et al.*, in press). Apparently, under nutrient repletion *Scenedesmus* try to avoid ingestion by the formation of large colonies in the presence of *Daphnia* (Hessen and Van Donk, 1993; Lampert *et al.*, 1994; Lürling and Van Donk, 1996). However, *Daphnia* induced colony formation was only observed in actively growing cells. Under nutrient depletion no colonies were induced. Nevertheless, under nutrient depletion grazing pressure on the limited algae is reduced as well directly by lower clearance rates and on the longer term by reduced *Daphnia* population growth.

CONCLUSIONS

Zooplankton growth is not only determined by the quantity of the algal food, but also by the food quality. Moreover, algae seem not defenceless particles easily harvested by zooplankton. They have evolved several strategies that reduce mortality under both nutrient-replete and deplete conditions, which affect the energy-transfer from algae to herbivorous zooplankton.

ACKNOWLEDGEMENT

This study was supported by a grant from the Foundation of Life Sciences (SLW-grant 805-37-361) of the Dutch organisation for Scientific Research (NWO).

REFERENCES

Ahlgren G., Lundstedt L., Brett M. and Forsber C. (1990). Lipid composition and food quality of some freshwater phytoplankton for cladoceran zooplankters. *J. Plankton Res.*, **12**, 809-818.

Burns C.W. (1968). The relationship between body size of filter-feeding Cladocera and the maximum size of particles ingested. *Limnol. Oceanogr.*, **13**, 675-678.

Butler N.M., Suttle C.A. and Neill W.E. (1989). Discrimination by freshwater zooplankton between single algal cells differing in nutritional status. *Oecologia*, **78**, 368-372.

Canter-Lund H and Lund J.W.G. (1995). *Freshwater Algae. Their microscopic world explored.* Biopress Ltd., 360 pp.

DeMott W.R. (1986). The role of taste in food selection by freshwater zooplankton. *Oecologia*, **69**, 334-340.

Giani A. (1991). Implications of phytoplankton chemical composition for zooplankton production: experimental evidence. *Oecologia*, **87**, 409-416.

Golterman H.L. and Clymo R.S. (eds.) (1969). *Methods for Chemical Analysis of Fresh Waters*, IBP Handbook, **8**, Blackwell, pp 172.

Groeger A.W., Schram M.D. and Richard G. (1991). Influence of food quality on growth and reproduction in *Daphnia. Freshwater Biol.*, **26**, 11-19.

Hassid W.Z. and Abraham S. (1957). Chemical procedures for analysis of polysaccharides. I. Determination of glycogen and starch - Determination of glycogen with anthrone reagent. In: Golowick S.P. and Kaplan N.O. (eds.) "*Methods in Enzymology*", **3**, Academic Press NY, 35-36.

Hessen D.O. (1992). Nutrient element limitation of zooplankton production. *Am. Nat.*, **140**, 799-814.

Hessen, D.O. and Van Donk, E. (1993). Morphological changes in *Scenedesmus* induced by substances released from *Daphnia. Arch. Hydrobiol.*, **127**, 129-140.

Lampert W. (1987). Feeding and nutrition in *Daphnia*. In: Peters R.H. and De Bernardi R.(eds.), "*Daphnia*" *memorie dell'Istituto Italiano di Idrobiologia*, **45**, 143-192.

Lampert, W., Rothhaupt, K.O. and von Elert, E. (1994). Chemical induction of colony formation in a green alga (*Scenedesmus acutus*) by grazers (*Daphnia*). *Limnol. Oceanogr.*, **39**, 1543-1550.

Lürling, M. and Van Donk, E. (1996). Zooplankton-induced unicell-colony transformation in *Scenedesmus acutus* and its effect on growth of herbivore *Daphnia. Oecologia*, **108**, 432-437.

Lowry O.H., Rosebrough W.J., Farr A.L. and Randall R.J. (1951). Protein measurement with Folin phenol reagent. *J.Biol. Chemistry*, **193**, 265-275.

Lynch M. (1989). The life-history consequences of resource depression in *Daphnia pulex*. *Ecology*, **70**, 246-256.

Meyer J.S., Ingersoll C.G., McDonald L.L. and Boyce M.S. (1986). Estimating uncertainty in population growth rates: jack-knife vs. bootstrap techniques. *Ecology*, **67**, 1156-1166.

Meyer E. and Walther A. (1988). Methods for the estimation of protein, lipid, carbohydrate and chitin levels in fresh water invertebrates. *Arch. Hydrobiol.*, **113**, 161-177.

Mitchell S.F., Trainor F.R., Rich P.H. and Goulden C.E. (1992). Growth of *Daphnia magna* in the laboratory in relation to the nutritional state of its food species, *Chlamydomonas reinhardtii*. *J. Plankton Res.*, **14**, 379-391.

Müller-Navarra D.C. (1995). Biochemical versus mineral limitation in *Daphnia. Limnol. Oceanogr.*, **40**, 1209-1214.

Porter K.G. (1975). Viable gut passage of gelatinous green algae ingested by *Daphnia. Verh. Int. Ver. Theor. Angew. Limnol.*, **19**, 2840-2850.

Riegman, R. (1985). Phosphate-phytoplankton interactions. *PhD-thesis*. Univ. of Amsterdam. pp.143.

Rothhaupt K.O. (1995). Algal nutrient limitation affects rotifer growth rate but not ingestion rate. *Limnol. Oceanogr.*, **40**, 1201-1208.

Ryther J.H. (1954). Inhibitory effects of phytoplankton upon the feeding of *Daphnia magna* with reference to growth, reproduction and survival. *Ecology*, **35**, 522-533.

Skulberg O.M. and Skulberg R. (1990). Research with algal cultures. NIVA's culture collection of algae. *NIVA-report* ISBN 82-551743-6.

Sommer, U. (1989). The role of competition for resources in phytoplankton succession. In: Sommer U. (ed.) *Plankton ecology: Succession in plankton communities*, Springer, 57-106.

Sterner R.W. (1989). The role of grazers in phytoplankton succession. In: Sommer U. (ed.) *Plankton ecology: Succession in plankton communities*, Springer, 107-170.

Sterner R.W. (1993). *Daphnia* growth on varying quality of *Scenedesmus*: mineral limitation of zooplankton. *Ecology*, **74**, 2351-2360.

Sterner R.W., Hagemeier D.D., Smith W.L. and Smith R.F. (1993). Phytoplankton nutrient limitation and food quality for *Daphnia*. *Limnol. Oceanogr.*, **38**, 857-871.

Sterner R.W. and Smith, R.F. (1993). Clearance, ingestion and release of N and P by *Daphnia obtusa* feeding on *Scenedesmus acutus* of varying quality. *Bull. Mar. Sci.*, **53**, 228-239.

Trainor F.R. (1991). Discovering the various ecomorphs of *Scenedesmus*. The end of a taxonomic era. *Arch. Protistenkd.*, **139**, 125-132.

Urabe J. and Watanabe Y. (1992). Possibility of N and P limitation for planktonic cladocerans: an experimental test. *Limnol. Oceanogr.*, **37**, 244-251.

Van Den Hoek C., Mann D.G. and Jahns H.M. (1995) *Algae: An introduction to phycology*. Cambridge, 623 pp.

Van Donk E. & Hessen D.O. (1993). Grazing resistance in nutrient-stressed phytoplankton. *Oecologia*, **93**, 508-511.

Van Donk E., Lürling M., Hessen D.O. and Lokhorst B. (1997). Changed cell wall morphology in nutrient deficient phytoplankton and the significance for grazers. *Limnol. Oceanogr.*, **42**, 357-364.

Van Donk, E., Lürling, M. and Lampert, W. (in press). Consumer-induced changes in phytoplankton: inducibility, costs, benefits and impacts on grazers. *In* Harvell, D. and Tollrian, R. [eds.] *Consequences of inducible defenses for population biology*. Princeton University.

CYANOBACTERIAL DOMINANCE IN LAKE VELUWEMEER AND LAKE WOLDERWIJD, THE NETHERLANDS

H.H. Reeders [1], P.C.M. Boers [2], D.T. van der Molen [2], T.H. Helmerhorst [1]

[1] *Dutch Water Authority Directorate IJsselmeer Area, P.O. Box 600, 8200 AP Lelystad, the Netherlands*
[2] *Institute for Inland Water Management and Waste Water Treatment, P.O. Box 17, 8200 AA Lelystad, the Netherlands*

ABSTRACT

Lake Veluwemeer and Lake Wolderwijd have long suffered from eutrophication stress. Recently, however, water quality has improved substantially. An overview of the developments in water quality (transparancy, P_{tot}, chlorophyll) is presented for 1969-1996. Dominance of filamentous cyanobacteria in summer is analyzed. P-reduction is the key-factor to reduce cyanobacterial dominance. At P_{tot}>0.20 mg l^{-1} cyanobacterial dominance approaches 100%, at 0.05<P_{tot}<0.20 mg l^{-1} the dominance decreases rapidly, while at P_{tot}≤0.05 mg l^{-1} filamentous cyanobacteria virtually disappear. In the intermediate range for P summer dominance of cyanobacteria may be either high or low. The occurrence of these two states is related to the winter conditions and the global insolation in spring (April-May). Low summer dominances may occur after a severe winter. A severe winter results in a low inoculum of filamentous cyanobacteria, a mild winter in a high inoculum. High cyanobacterial dominance in summer is related to a high inoculum and/or high global insolation in spring. When the inoculum is low, i.e. after a severe winter, cyanobacterial dominance in summer may be either low or high. Low global insolation does not provide an indication for poor conditions for growth. It is suggested that good conditions for growth in spring determine cyanobacterial growth more than poor conditions.

KEYWORDS

Algae; cyanobacteria; *Oscillatoria*; eutrophication; nutrients; phosphorus; water quality; lakes; Lake Veluwemeer; Lake Wolderwijd.

INTRODUCTION

The polder-borderlakes are situated between the land reclamations in Lake IJsselmeer and the "old" land (Fig. 1). Being shallow and enclosed systems these lakes are prone to eutrophication. For decennia cyanobacterial blooms, mainly *Oscillatoria* sp., have held the lakes in a tight grip, but recently a remarkable recovery is taking place: cyanobacterial dominance is broken, transparancy has improved, macrophyte and *Chara* cover increases every year, waterfowl numbers show a dramatical increase and biodiversity has gone up (Coops et al., 1997; Noordhuis et al., 1997).

Breaking cyanobacterial dominance is crucial for ecological recovery of lakes under eutrophication stress. There are different views on what determines cyanobacterial dominance. Jensen *et al.* (1994) and Schreurs (1992) consider the P-concentration to be the key-factor. Scheffer *et al.* (1997), however, could not find a relation with P, but stressed the importance of turbidity (shade) for explaining cyanobacterial dominance. Winter effects have been suggested by various authors (e.g. Berger, 1975; Hosper, 1997), but these have never been confirmed statistically.

In this article the factors determining cyanobacterial dominance in summer are analyzed for the situation in Lake Veluwemeer (3350 ha, mean depth 1.5 m) and Lake Wolderwijd (1850 ha, mean depth 1.8 m). In these lakes several measures to improve water quality have been taken and an extensive database of water quality parameters is available since 1969. A better understanding of cyanobacterial dominance is wanted in relation to the measures taken and external factors that can not manipulated, like weather conditions.

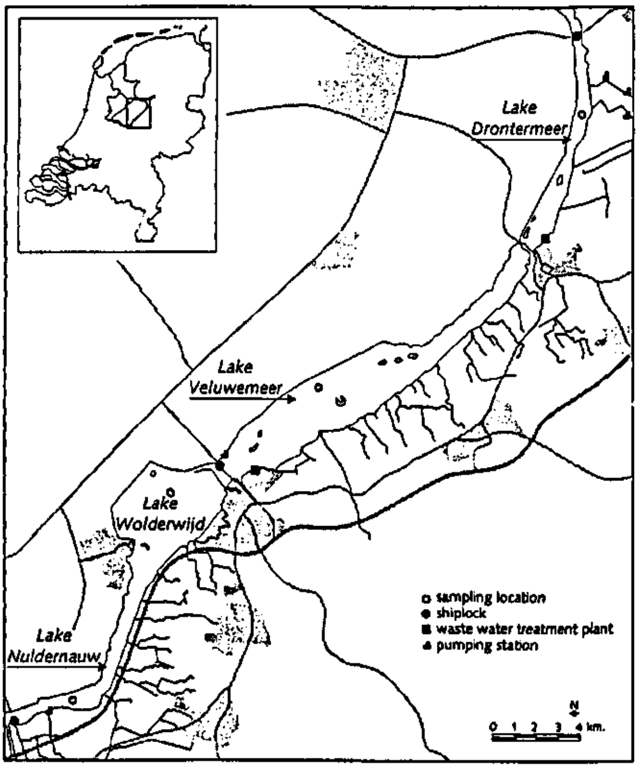


Figure 1. The polder-borderlakes in the Netherlands.

WATER QUALITY IN LAKE VELUWEMEER AND LAKE WOLDERWIJD

Three measures to abate eutrophication have been taken so far: 1) flushing, 2) phosphorus stripping at two waste water treatment plants and 3) biomanipulation (fish stock reduction). The main effects of flushing are dilution and phosphorus inactivation in the sediment (Jagtman *et al.*, 1992; Hosper, 1997). In Lake Veluwemeer the internal phosphorus loading decreased by a factor 7 between 1978 and 1987 due to the oxidizing action of nitrate on the sediment and due to fixation with calcium (Jagtman *et al.*, 1992). The hydraulic residence time has been reduced from >1 year to 2-3 months (Hosper, 1997). Phosphorus stripping is being practised in the waste water treatment plants of Elburg, discharging its effluent on Lake Drontermeer, and Harderwijk, discharging on Lake Veluwemeer, since 1972 and 1979 respectively. Biomanipulation was carried out in Lake Wolderwijd in winter 1990-1991 (Meijer & Hosper, 1997). The fishstock was reduced by 75%. The operation was the largest of this kind known (Grimm & Backx, 1994). The measures resulted in an improved water quality. Fig. 2 shows the development in summer-average (April-September) P_{tot} concentration, chlorophyll-a concentration (chlorophyll-a data before 1985 were corrected according to Hosper, 1997) and transparancy (Secchi) from 1969-1996.

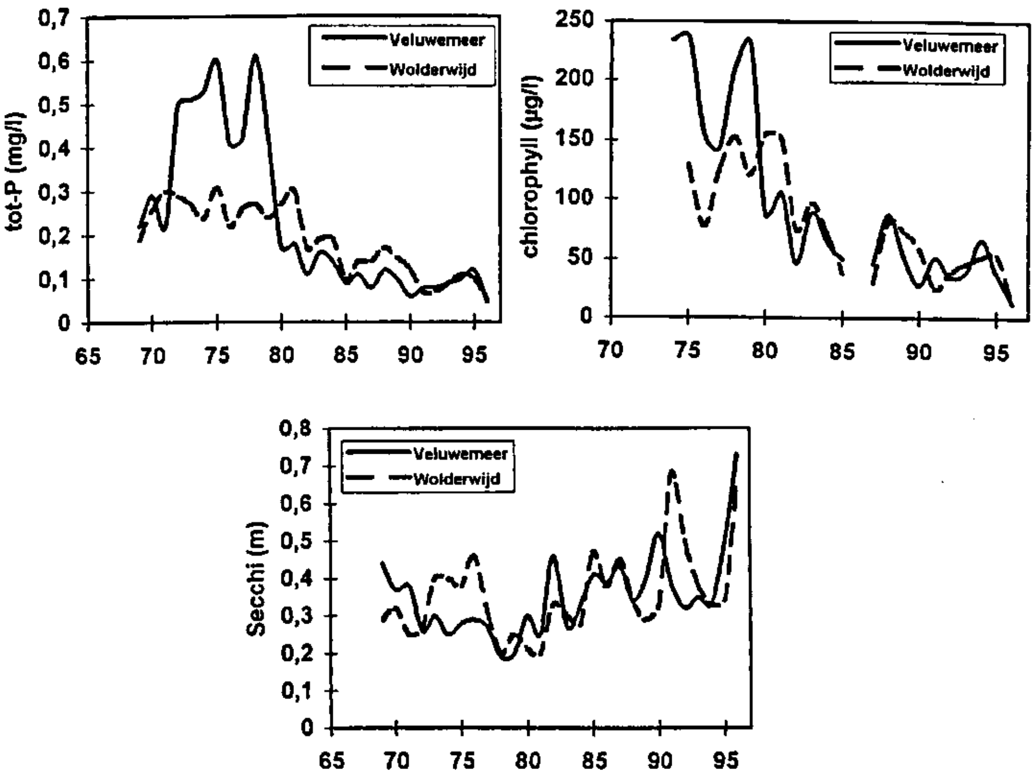


Figure 2. Summer-average (April-September) concentrations of P_{tot}, chlorophyll-a and transparancy (Secchi) in Lake Veluwemeer and Lake Wolderwijd, the Netherlands, 1969-1996.

The P_{tot}-levels were high in the 1970's, especially in Lake Veluwemeer. P_{tot} decreased substantially in 1980 in Lake Veluwemeer as a result of phosphorus stripping at the waste water treatment plant in combination with flushing. Lake Wolderwijd showed a delay in decrease due to a later start of flushing operations (1981). P_{tot} continued to decrease to 0.05 mg P l^{-1} in 1996, the lowest summer average value so far recorded in both lakes. Chlorophyll-a neatly follows P_{tot}, with an exceptionally low summer average value of 13 µg l^{-1} in 1996 in both lakes. Dominance by filamentous cyanobacteria was completely broken in 1996. *Oscillatoria* virtually disappeared from the plankton spectrum, while non-filamentous species like *Merismopedia* and *Cyanodiction* emerged. Despite the strong decrease in chlorophyll-a, transparancy has improved only slightly. Transparancy increased from ± 0.2 m in 1979 to a level of 0.7 m in 1996. Transparancy showed a peak In Lake Wolderwijd in 1991 after biomanipulation. Most likely the measure has stimulated the development of *Chara*. Above the *Chara* meadows the water is crystal clear, a phenomenon observed in Lake Veluwemeer as well (Meijer & Hosper, 1997; van den Berg *et al*, in press). The time series (Fig. 2) of measurements on the routine sampling location of course does not reveal the heterogenous pattern of turbid and extremely clear patches in the lake.

A CONCEPTUAL MODEL FOR CYANOBACTERIAL DOMINANCE IN SUMMER

Excessive growth of filamentous cyanobacteria impairs ecosystem functioning: the turbid water prevents development of macrophytes and the fish community characteristically becomes dominated by only one or a few species. Hence, breaking the dominance of filamentous cyanobacteria has high priority for the lake manager (the Dutch Water Authority). Cyanobacterial dominance can be expressed as a percentage of total phytoplankton biovolume. Fig. 3a shows the relation between the summer average P_{tot}-concentration and cyanobacterial dominance in summer. P becomes limiting at ± 0.2 mg l^{-1}. Below this level the cyanobacterial dominance decreases rapidly. At P_{tot} > 0.2 mg l^{-1} cyanobacterial dominance approaches 100%, while at $P_{tot} \leq 0.05$ mg l^{-1} filamentous cyanobacteria disappear. Zooming in on the intermediate range (0.05-0.20 mg l^{-1}) reveals that cyanobacterial dominance may be either high (>50%) or low (<50%) at similar P_{tot}-concentrations (Fig. 3b). There is a relation with the winter conditions. After a severe winter (>50 days ice-cover) cyanobacterial dominance in summer is broken in half of the cases. A mild winter (<50 days ice-cover), however, is followed by a high dominance of cyanobacteria in summer ($p < 0.001$; binomial test).

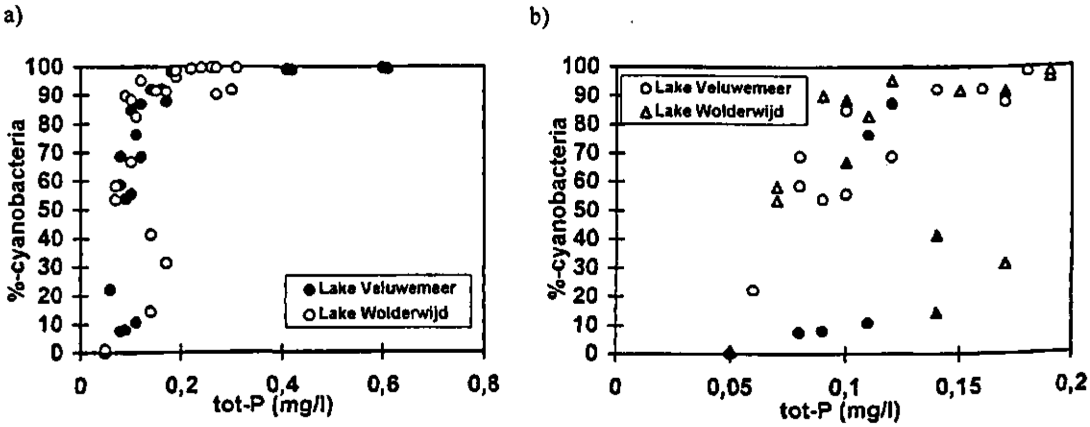


Figure 3. The relation between the summer average (April-September) P_{tot}-concentration and cyanobacterial dominance (%-biovolume) in Lake Veluwemeer and Lake Wolderwijd: a) all data, b) for $0.05<P_{tot}<0.20$ mg l^{-1}. Years with a severe winter (>50 days ice-cover) shaded black (b).

The above observations result in a conceptual model for growth of filamentous cyanobacteria *in the intermediate range for P_{tot}*, presented in Fig. 4. In the model, growth starts with a low (models 1 & 2) or high (models 3 & 4) inoculum (i.e. the biomass immediately prior to the growing season) and may result in a low or high summer dominance following poor (models 1 & 3) viz. good (models 2 & 4) growing conditions in spring (March-May).

Three hypotheses will be tested:

1. winter conditions determine the inoculum: after a severe winter the cyanobacterial inoculum is low and vice versa
2. summer dominance of cyanobacteria is related to the inoculum
3. summer dominance of cyanobacteria is related to conditions for growth in spring (March-May)

For testing hypothesis 3, the following parameters indicating for conditions for growth in spring were investigated: flushing rate, water temperature and global insolation. In March-May the flushing rate ranged between 0.0003 - 0.012 d^{-1}, water temperature between 8.5 - 13 °C and global insolation between 9 - 14.5 J m^{-2}. For further analysis the data were classified into two classes indicating good and poor conditions for growth respectively (see Table 1). "Good" is defined as: a low flushing rate (< 0.004 d^{-1}), a high water temperature (> 10 °C) or a high insolation (> 10 J m^{-2}). "Poor" is defined as: a high flushing rate (> 0.004 d^{-1}), a low water temperature (< 10 °C) or low insolation (< 10 J m^{-2}).

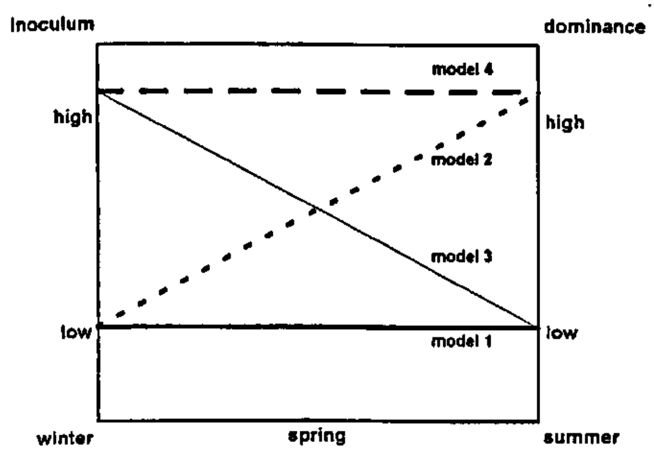


Figure 4. A conceptual model for the dominance of filamentous cyanobacteria in summer.

Table 1. Classification of winter conditions, cyanobacterial inoculum, cyanobacterial dominance in summer, and factors indicating for growing conditions (flushing rate, water temperature, insolation) in relation to the expected conceptual models (N.B.: summer average P_{tot} in the range $0.05<P_{tot}<0.20$ mg l^{-1}). n.d. = no data. VM = Lake Veluwemeer. WW = Lake Wolderwijd

Class	winter	inoculum	flushing rate	temperature	insolation	summer dominance
1	mild	> 10 mm^3/ml	< 0.004 d^{-1}	> 10 °C	> 10 j/m$_2$	high (> 50%)
2	severe	< 10 mm^3/ml	> 0.004 d^{-1}	< 10 °C	< 10 J/m^2	low (< 50%)

year	winter	inoculum		summer dominance		model expected		flushing rate		water temperature		insolation
		VM	WW	VM	WW	VM	WW	VM	WW	VM	WW	VM/WW
1980	1	2	-	1	-	2	-	2	-	1	1	1
1981	1	1	-	1	-	4	-	1	1	1	1	1
1982	2	2	nd	1	1	2	2/4	1	2	2	2	1
1983	1	1	1	1	1	4	4	1	2	2	2	1
1984	1	2	1	1	1	2	4	1	1	2	2	1
1985	2	2	1	2	1	1	4	2	1	1	2	2
1986	2	2	2	2	2	1	1	1	1	1	1	1
1987	2	2	2	2	2	1	1	2	1	2	2	1
1988	1	2	2	1	2	2	1	1	1	1	1	1
1989	1	2	1	1	1	2	4	2	1	1	1	1
1990	1	1	1	2	1	3	4	1	1	1	1	1
1991	1	2	1	1	1	2	4	2	1	1	1	1
1992	1	2	1	1	1	2	4	2	2	1	1	1
1993	1	nd	nd	1	1	2/4	2/4	1	2	1	1	1
1994	1	nd	nd	1	1	2/4	2/4	2	2	2	2	2
1995	2	nd	nd	1	1	2/4	2/4	2	2	2	1	2

CYANOBACTERIAL DOMINANCE IN SUMMER: RESULTS OF TESTING THE HYPOTHESES

Table 2 shows the results of the statistical analysis of the three hypotheses, based on the information in Table 1.

Table 2. Statistical analysis of the relation between cyanobacterial inoculum and winter conditions viz. summer dominance of cyanobacteria, and between cyanobacterial dominance and flushing rate viz. water temperature viz. global insolation.
* = significant ($P \leq 0.10$). ** = highly significant ($P \leq 0.05$). ns = not significant.

	inoculum		
	high (class 1)	low (class 2)	binomial test
winter conditions			
mild (class 1)	9	7	n.s.
severe (class 2)	1	6	p = 0.062 *
	summer dominance		
	high (class 1)	low (class 2)	binomial test
inoculum			
high (class 1)	9	1	p = 0.011 **
low (class 2)	7	6	n.s.
confirmation/rejection scores for predicted growth models			
	confirmation	*rejection*	binomial test
env. parameter			
flushing rate	13	17	n.s.
water temperature	15	15	n.s.
insolation	19	11	p = 0.10 *

The type of winter (severe/mild) and the inoculum of filamentous cyanobacteria (biomass prior to growing season) are clearly related. There is a significant ($p = 0.062$) relation between a severe winter and a low inoculum; only one case does not comply: Lake Wolderwijd, 1985. After a mild winter, however, the inoculum may be either high or low with approximately equal probability.

At a low inoculum the summer dominance of cyanobacteria may be either high or low. A high inoculum, however, is strongly related to high summer dominance ($p = 0.011$). Only one case, Lake Veluwemeer 1990, does not comply. Hence, a high summer dominance can follow after both a high or a low inoculum, but a low summer dominance is related to a low inoculum ($p=0.062$).

No relation could be found between flushing rate or water temperature and the predicted conditions for growth, neither as a single factor nor in combination. Global insolation shows a 63% confirmation of the predicted conditions for growth in the models ($p = 0.10$). High global insolation confirms 75% of the predicted "good" growing conditions in models 2/4 ($p < 0.01$; binomial test). Vice versa, however, low global insolation does not predict for "poor" conditions for growth in models 1/3 (17% confirmation).

DISCUSSION

The efforts to improve water quality of Lake Veluwemeer and Lake Wolderwijd have started to pay off. The reduction of the external phosphorus loading has been crucial for the restoration proces. Analysis of cyanobacterial dominance in summer shows that P-reduction remains the key-factor for control: at a P_{tot}-concentration ≤ 0.05 mg l^{-1} filamentous cyanobacteria entirely disappear. For lakes with another morphometry and/or climatic conditions this level may be different (Schreurs, 1992; Jensen *et al.*, 1994;

Scheffer *et al.*, 1997), but the response is similar.

This study reveals that in the relation with summer dominance of filamentous cyanobacteria three ranges in P_{tot}-concentration can be identified: 1) high P levels (P_{tot}> 0.20 mg l^{-1}) whereby P is not limiting and cyanobacterial dominance approaches 100%, 2) intermediate P levels (0.05<P_{tot}<0.20 mg l^{-1}) whereby P is limiting and filamentous cyanobacteria show a quick decline with decreasing P_{tot}-levels, and 3) low P levels (P_{tot}≤0.05 mg l^{-1}) whereby filamentous cyanobacteria disappear. The intermediate P-range is comparable to the range identified by Schreurs (1992), who found that cyanobacterial dominance is broken abruptly between 0.05-0.15 mg l^{-1} P. The narrow P_{tot}-range of response may be the explanation why Scheffer *et al.* (1997), looking at a much wider range, could not find a relation between P and cyanobacterial dominance. A close look on their data reveals, however, that none of the 55 lakes investigated showed cyanobacterial dominance at P_{tot}≤0.05 mg l^{-1}.

Winter effects on cyanobacterial dominance were only identified in the intermediate P-range. No winter effects were identified at high P_{tot}-levels. Statistical evidence has been provided that a severe winter results in a low inoculum, while a mild winter may result in either a low or high inoculum. A high inoculum is related to a high summer dominance of cyanobacteria. A low inoculum, however, does not necessarily result in a reduced population of filamentous cyanobacteria in summer: high or low summer dominance are still equally possible. This can at least partly be attributed to the conditions for growth in spring, notably the global insolation. When the global insolation is high, the summer dominance will be high. Obviously, even the higher light intensities in spring do not provoke photo-inhibition, as may be expected for filamentous cyanobacteria (Van Liere & Walsby, 1982). The reverse is not the case: there is no correlation between low global insolation and low summer dominance. This may be explained by the fact that *Oscillatoria* species are known to be good competitors for light at low intensities. Scheffer *et al.* (1997) stress the importance of turbidity (shade) as the factor determining cyanobacterial dominance. Schreurs (1992) showed that the frequency of cyanobacterial dominance is less with high light conditions.

Flushing rate and water temperature in spring failed to confirm predicted conditions for growth of filamentous cyanobacteria. The flushing rate is far too small to wash out phytoplankton considering the prevailing temperatures. At temperatures > 2 °C net growth of cyanobacteria will already offset the effects of flushing at the operational rates (Jagtman *et.al.*, 1992). At an average temperature in March-May of ± 10 °C the flushing rate should be ± 0.3 d^{-1} to have any effect, which is an almost riverine situation not applicable to the situation in the borderlakes.

The observations coincide with the concept of alternative equilibrium states in an intermediate range of P put forward by Scheffer *et al.* (1997). Severe winter conditions then may be seen as the trigger to shift from high to low cyanobacterial dominance, while after mild winters, resulting in a high inoculum, high dominance will persist.

The analysis shows the importance of "good" conditions for growth in spring (i.e. high insolation) for cyanobacterial growth, but indicators for "poor" growing conditions (i.e. low insolation) could not be identified. Although the present approach arrives at a satisfactory explanation for high summer dominance of cyanobacteria, low summer dominance can only be explained partly. Of course the selection of factors has not been limitative while the abstraction level of analysis only permits a correlative study. Adding variables and analysis in higher detail would be required to enhance the level of prediction, particularly for low summer dominance.

CONCLUSIONS

1. P-reduction is the key-factor to break cyanobacterial dominance
2. three ranges in P_{tot} with respect to summer dominance of filamentous cyanobacteria are identified:
   $P_{tot}>0.20$ mg l^{-1} : dominance approaches 100%
   $0.05<P_{tot}<0.20$ mg l^{-1} : dominance declines abruptly
   $P_{tot}\leq0.05$ mg l^{-1} : disappearance of filamentous cyanobacteria
3. winter effects on cyanobacterial dominance occur in the intermediate P-range: a severe winter followed by low dominance in summer
4. a severe winter results in a low inoculum, a mild winter in a high inoculum of filamentous cyanobacteria
5. cyanobacterial dominance in summer is high when the inoculum is high and/or when the global insolation in spring is high
6. when the inoculum is low, cyanobacterial dominance in summer can be low or high
7. good conditions for growth in spring appear to be more important for filamentous cyanobacteria than poor conditions for growth

ACKNOWLEDGEMENTS

A huge number of people have been sampling and doing chemical and biological measurements to fill the database on water quality since 1969. We acknowledge their effort and dedication through which we were enabled to make the analysis.

REFERENCES

Berg, M.S. van den, Coops, H., Meijer, M.-L., Scheffer, M. & Simons J. (*in press*). Clear water associated with a dense *Chara* vegetation in the shallow and turbid lake Veluwemeer, The Netherlands. In: *The role of submerged macrophytes in structuring the biological community and bio-chemical dynamics in lakes*, E. Jeppesen & M. Sondergaard (Eds.).

Berger, C. (1975). Occurrence of *Oscillatoria agardhii* Gom. in some shallow eutrophic lakes. *Verhandlungen der Internationale Vereinigung für theoretische und angewandte Limnologie*, 19, 2689-2697.

Coops, H., Doef, R.W., Witte, B.J. de & van den Berg, M.S. (1997). Recovery of aquatic vegetation in the Lake IJsselmeer region. *De Levende Natuur*, 98(1), 8-13. (in Dutch, English summary).

Grimm, M.P. & Backx, J.J.G.M. (1994). Mass removal of fish from Lake Wolderwijd, the Netherlands. Part I: planning and strategy of a large-scale biomanipulation project. In: *Rehabilitation of freshwater fisheries*, I.G. Cowx (Ed.), Fishing News Books, 390-400.

Hosper, S.H. (1997). *Clearing lakes. An ecosystem approach to the restoration and management of shallow lakes in the Netherlands.* Ph.D. Thesis, Agricultural University, Wageningen, the Netherlands.

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

Jensen, J.P., Jeppesen, E., Olrik K. & Kristensen, P. (1994). Impact of nutrients and physical factors on the shift from cyanobacterial to chlorophyte dominance in shallow Danish lakes. *Can. J. Fish. Aquat. Sci.*, 51, 1692-1699.

Liere, L. van & Walsby, E.A. (1982). Interactions of cyanobacteria with light. In: *The biology of cyanobacteria*, N.G. Carr & B.A. Whitton (Eds.), Blackwell, Oxford, UK.

Meijer, M.-L. & Hosper, S.H. (1997). Effects of biomanipulation in the large and shallow Lake Wolderwijd, the Netherlands. *Hydrobiologia*, 342/343, 335-349.

Noordhuis, R., van Roomen, M.W.J., Zollinger, R., Tempel, J. & Bouw, W. (1997). Historical perspectives on the waterbirds in the Randmeren-area. *De Levende Natuur* 98(1), 25-34 (in Dutch, English summary).

Scheffer, M., Rinaldi, S., Gragnani, A., Mur, L.R. & van Nes, E. (1997). On the dominance of filamentous cyanobacteria in shallow, turbid lakes. *Ecology*, 78(1), 272-282.

Schreurs, H. (1992). *Cyanobacterial dominance, relations to eutrophication and lake morphology.* Ph.D. Thesis, University of Amsterdam. the Netherlands.

Siegel, S. & Castellan, N.J. (1988). *Nonparametric Statistics for the Behavioural Sciences.* Second Edition. McGraw Hill.

THE BENTHONIC EUTROPHICATION IN THE MANAGEMENT OF SHALLOW LAKES AND WETLANDS

Gyula Lakatos, Mariannna Kiss and Magdolna K. Kiss

Ecological Department of L. Kossuth University, H-4010 Debrecen, P.O. Box 71, Hungary

ABSTRACT

Eutrophication, is a biological response to increased trophity, which is always caused by the enrichment of inorganic plant nutrients in water bodies. In order to act expediently and effectively against eutrophication, it is necessary to explore the sub-processes on the level of primary research. When the sub-processes of eutrophication in Hungary are concerned, a typical problem is constituted by the clarification of the role of submerged aquatic macrophytes. Therefore the present study gives an account of investigation results obtained in shallow lakes and wetlands with the purpose of revealing the trophic state and the role of the benthonic plant complex. Considering the biomass and chlorophyll *a* content of phytoplankton and submerged aquatic plants, it seems to be useful to distinguish between planktonic eutrophication (turbid water) and benthonic eutrophication (clear water) as corresponding to the locality of primary producers with their stabilizing, utilizing and circulating energy. As these differences are recognized, the problems of the trophity degree in the examined shallow lakes and wetlands can be more easily solved. Planktonic eutrophication is caused by microscopic algae in the open water that lacks substrate, while benthonic eutrophication is brought about by aquatic plants (submerged macrophytes, large multicellular filamentous algae, phytotecton).

KEYWORDS

Eutrophication; submerged aquatic macrophytes; shallow waterbodies; wetlands; planktonic eutrophication; benthonic eutrophication, biomanipulation.

INTRODUCTION

From today's anthropogenic impacts on the waters of Hungary, eutrophication is the most serious and general one (Vollenweider and Kerekes, 1980). Eutrophication is a biological response to increased trophity, which is always caused by the enrichment of by inorganic plant nutrients in water bodies (Mason, 1996). In shallow waterbodies, eutrophication is a natural succession process and its most apparent feature is the intensive growth of algae and aquatic plants in these ecosystems. The natural eutrophication process (Johnson and Vallentyne, 1971) usually comes with ageing of the shallow waterbodies, and can be accelerated by an increased input of nutrients. Cultural eutrophication takes place when this increase is due to human activities, and this state of waters can be disadvantageous from the aspect of water management and utilization. In order to act expediently and effectively against the eutrophication, it is necessary to explore the sub-processes on the level of primary research (Harper, 1992).

As the sub-processes of eutrophication in Hungary is concerned, a typical problem is constituted by the clarification of the role of submerged aquatic macrophytes. As opposed to the deep waterbodies, the benthonic plant complex (i.e. submerged macrophytes, filamentous and/or multicellular algae and attached epiphytic algae) should always be expected in the shallow waterbodies and wetlands of Hungary, and its role in eutrophication should be the subject of any comprehensive examination. Thus the following shallow waters of Hungary are suitable for investigation: the shallow Lake Velencei and wetlands, such as Darvas-Marsh (HNP), Rakamaz-Deadarm (in the flood plain of River Tisza) and a constructed wetland (NyPP).

This present study gives an account of the investigation results obtained in a shallow lake and wetlands with the purpose of revealing the trophic state and the role of the benthonic plant complex. For the trophity degree in different waters (Felföldy, 1976), the chlorophyll *a* contents of the phytoplankton and benthonic plant complex were examined simultaneously. Besides, the biomass, as well as, the phosphorus and nitrogen contents of submerged aquatic plants were measured.

METHODS

Situated near Budapest, Lake Velencei is one of the largest sodic, eutrophic shallow lake in Central Europe with a surface area of 26 km^2 and mean depth of 1.2 m. The water with high nutrient content enters the lake via the Császár Stream from the north-west, and the effluent is quite close to the entering site. A characteristic feature of the shallow lake is its mosaic-like surface, which is divided into several water regions by the reed islands and reed walls of common reed (*Phragmites australis*), and the clearings (= "tisztás") are only connected to these water regions by thick reed stands. In this way, the water regions have different water quality. On the basis of the chemical and biological examinations (Lakatos, 1978), the following water areas can be distinguished:
- dark-brown water area (Hollós-tisztás, Hosszú-tisztás, Nagytó-Rigya, Vendel-tisztás)
- grey water area (Nagy-tisztás)
- algal-brown water area (Kárászos, Öreg-tisztás)
- green water area (Fürdető)

Almost 20 % of the open surface of Lake Velencei has been covered with submerged aquatic plants, and the dominant benthonic plant was a filamentous alga, *Vaucheria dichotoma* (Lakatos, 1976). Thorough investigations were carried out on the submerged aquatic plants in a transect at Hosszú-tisztás, and the *Vaucheria* communities of Hollós-tisztás and Nagytó-Rigya were also subjected to sampling. Phytoplankton and aquatic macrophyte samples were collected in the summer of 1973.

Lying 5 kilometres from the village of Nagyiván, Darvas Marsh is a member of the Kunkápolnás Marshes in the Hortobágy National Park. Including the reed belt, the surface of this natural wetland is 12 hectares, and the mean depth of the water is 0.7 metre. Here, water is supplied by rain and spring thaws, although recently the marsh has received water from River Tisza via the local canal system, which caused changes in the sodic character of the marsh (Lakatos and Kiss, 1983).

Investigations have been performed on the water quality, phytoplankton and submerged aquatic plants since 1975 in order to explore the changes in the trophity state and their effects on the marsh, itself. The present study, however, only reports on the 1981 results, when the examinations were focused on the characteristic plant communities along a permanent transect of the marsh.

Rakamaz-Deadarm can be found in the outer part of the flood plain of River Tisza near the town of Tokaj, and it has an area of 90 hectares and mean depth of 1.8 m. Naturally, the occasional flooding of River Tisza provides the water to the deadarm via the summer dike, but rain water and spring thaws also play important roles in the water supply.

The largest part of the deadarm is covered with the communities of submerged (*Ceratophyllum demersum*, *Myriophyllum spicatum*), floating (*Nymphaea alba*, *Trapa natans*) and emergent (*Phragmites australis*, *Typha angustifolia*) plants. The high external loading of organic and inorganic materials have caused considerable siltation and rapid cultural eutrophication. In 1995, limnological investigations were carried out to evaluate the water quality, and to find applicable solutions for the biomanipulation treatment. Samples of water, mud, phytoplankton and aquatic macrophytes were taken from 20 sites of the deadarm.

The Nyírbogdány Petrochemical Plant has a post-treatment pond system (NE-Hungary). This constructed wetland was established in the early 1970s, was planted with rooted emergent macrophytes, has been operating as a free water surface system (Lakatos et al., 1997). The reed-submerged weeds pond has a surface area of 1.5 hectares and mean depth of 1 m, and it is daily loaded with 200-250 m^3 biologically treated petrochemical wastewater.

Investigations were performed four times during the vegetation periods for several years. Beside measuring the physical and chemical parameters of the water and mud, the chlorophyll *a* contents of phytoplankton and submerged aquatic plants (*Chara sp., Potamogeton pectinatus*) were determined, as well. Samples of aquatic macrophytes were collected for the determination of the element concentrations with ICP-AAS analysis, while the dry matter and ash contents were also measured.
The detailed description of the applied sampling and investigations methods can be found in Lorenzen (1967), Felföldy (1974) and Lakatos (1976) papers.

RESULTS AND DISCUSSION

In waters, organic materials are produced by the phytoplankton, aquatic macrophytes and periphytic phytotecton (Wetzel, 1964). Depending on the type of the surface standing waters (deep waterbodies, shallow waterbodies and wetlands), different primary producer organisms can play the dominant role due to a set of morphological, hydrological, physical, chemical and biological properties.

The estimation of the trophic state in waters can be carried out by observations on the intensity of phytoplankton, aquatic macrophytes and phytotecton (Schwoerbel, 1971). The estimation needs measurement on the quantity of aquatic plants, and thus on the intensity of primary production are necessary. Such a quantitative survey can be performed by the determination of the chlorophyll *a* and biomass expressed in dry weight for a certain volume or surface unit.

The data concerning the chlorophyll *a* and biomass of submerged aquatic plant communities on bottom units are given in Table 1. In the waterbodies, the chlorophyll *a* contents were also determined. The concentrations of chlorophyll *a* were regarded as indices for the biomass of algae and local aquatic plants, and together with the water transparency values (Secchi disc), they were used to classify the trophic states of the examines watersbodies (Mason, 1996).

Table 1. Data for biomass and chlorophyll *a* of submerged macrophyte communities in LakeVelencei and wetlands

sampling sites	year	plant community	dw	Chl *a*
			g/m^2	
Lake Velencei	1973			
Hosszú-tisztás I.		Myriophyllum	646	1.04
II.		Potamogeton	1080	1.67
III.		Vaucheria	734	1.25
Hollós-tisztás		Vaucheria	725	1.57
Nagytó-Rigya		Vaucheria	318	0.46
Darvas Marsh	1981	Vaucheria	800	0.61
		Myriophyllum	888	1.28
Rakamaz Deadarm	1995	Myriophyllum	414	1.30
		Ceratophyllum	518	1.59
Reed-submerged weed	1995			
unit - NyPP		Chara	271	1.00
		Potamogeton	89	0.33

In recent years, eutrophication has caused more and more problems that could be detected in phytoplanktonic overgrowth, as well as, in changes in the processes of aquatic ecosystems. From this empirical fact, Thomas (1968) concluded that in fact, every "manure dumping" that results in larger algal production is eutrophication. However, the findings in the shallow Lake Velencei, Darvas Marsh, Rakamaz Deadarm and the reed-submerged weeds pond (constructed wetland) of Nyírbogdány supports the assumption that eutrophication can not only be caused by the growth of phytoplanktonic biomass, but by intensive benthonic vegetation, as well. From the values in Table 2 and Fig. 1, it is clear that submerged aquatic plants must also be taken into consideration when determining the trophic state in shallow waterbodies and wetlands, since significant part of nutrients are utilized by these living organisms, so eutrophication is also brought about by them. A comparison between the benthonic phytomass values obtained from the biomass and chlorophyll *a* analyses with the literature data (Juday, 1924; Bernatowicz and Pieczynska, 1965; Hartman and Brown, 1967) supports the eutrophic character of the brown-water areas of Lake Velencei and the wetlands.

Table 2. The quantity of planktonic and benthonic chlorophyll *a* in Lake Velencei and wetlands

sampling sites	plant community	chlorophyll *a*			trophity index
		plankton	benthon mg/m^3	total	
Lake Velencei					
Hosszú-tisztás I.	Myriophyllum	7.5	870	877.5	9
II.	Potamogeton	13.6	1047	1060.6	9
III.	Vaucheria	13.1	638	651.1	8
Hollós-tisztás	Vaucheria	18.9	1566	1584.9	9
Nagytó-Rigya	Vaucheria	8.9	307	315.9	8
Darvas Marsh					
	Vaucheria	18.3	380	398.3	8
	Myriophyllum	13.5	1803	1816.5	9
Rakamaz Deadarm	Myriophyllum	8.2	676	684.2	8
	Ceratophyllum	10.4	1174	1184.4	9
Reed-submerged weed					
unit - NyPP	Chara	4.1	1004	1008.1	9
	Potamogeton	3.9	268	271.9	8

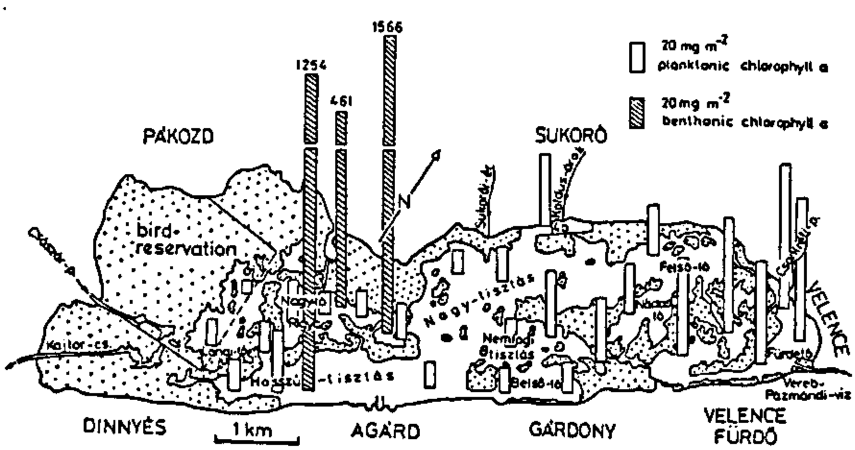


Fig. 1. The quantity of planktonic and benthonic chlorophyll content in the different parts of Lake Velencei (August 1973)

Considering the biomass and chlorophyll *a* content of phytoplankton and submerged aquatic plants, it seems to be useful to distinguish between planktonic eutrophication (turbid water) and benthonic eutrophication (clear water) as corresponding to the locality of primary producers with their stabilizing, utilizing and circulating energy. As these differences are recognized, the problems of the trophity degree in the examined shallow lake and wetlands can be more easily solved.

Planktonic eutrophication is caused by microscopic algae in the open water that lacks substrate, while benthonic eutrophication is brought about by aquatic plants (submerged macrophytes, large multicellular filamentous algae, phytotecton). The role of benthonic vegetation representing an immense biomass is important from the viewpoint of the "life" and the management of waters, primarily because it stabilizes large quantities of nutrients thus control the abnormal overgrowth of phytoplankton, and ensures the oxygen supply of water.

The planktonic and benthonic types of eutrophication have influenced the conception of the investigation in several ways, and - as it has been revealed - they can be applied in solving several practical problems. Considering human activities, the relationship between the two types of eutrophication is vitally important in recreational and nature conservation. The planktonic type is less favourable than the benthonic one, because in the case of the former type, the suspending phytoplankton can be separated with no simple method.

Benthonic eutrophication is more advantageous for the protection of our environment, because the submerged macrophytes and attached epiphytic periphyton remain in the water. Besides, the diversity of species often increases, and the benthonic vegetation provides appropriate habitats to the biota. In the light of these observations it seems to be reasonable to construct post-treatment pond systems with the last unit having submerged benthonic vegetation. In this way, the system would exercise biofiltering and nutrient eliminating effect (Reddy, 1983; Wolverton and McDonald, 1979; Irvine et al., 1989; Balls et al., 1989; Lakatos et al., 1997; etc.).

In our opinion, however, further investigations are required to establish the role of the benthonic vegetation in shallow waterbodies and wetlands more exactly.

CONCLUSIONS

From today's anthropogenic impacts on the waters of Hungary, eutrophication is the most serious and general one. In order to act expediently and effectively against eutrophication, it is necessary to explore the sub-processes on the level of primary research. When the sub-processes of eutrophication in Hungary are concerned, a typical problem is constituted by the clarification of the role of submerged aquatic macrophytes. As opposed to deep watersbodies, the benthonic plant complex (i.e. submerged macrophytes, filamentous and/or multicellular algae and attached epiphytic algae) should always be expected in the shallow waterbodies and wetlands of Hungary, and its role in eutrophication should be the subject of any comprehensive examination.

The findings in the shallow Lake Velencei, Darvas Marsh, Rakamaz Deadarm and the reed-submerged weed pond (constructed wetland) of Nyírbogdány supports the assumption that eutrophication can not only be caused by the growth of phytoplankton biomass, but by intensive benthonic vegetation, as well.

Considering the biomass and chlorophyll *a* content of phytoplankton and submerged aquatic plants, it seems to be useful to distinguish between planktonic eutrophication (turbid water) and benthonic eutrophication (clear water) as corresponding to the locality of primary producers with their stabilizing, utilizing and circulating energy. As these differences are recognized, the problems of the trophity degree in the examined shallow lake and wetlands can be more easily solved.

Benthonic type of eutrophication is more advantageous for the protection of our environment, because the submerged macrophytes and attached epiphytic periphyton remain in the water. Besides, the diversity of species often increases, and benthonic vegetation provides advantageous habitats to the biota. In the light of these observations, it seems to be reasonable to construct post-treatment pond systems with the last unit having submerged benthonic vegetation. In this way, the system would exercise biofiltering and nutrient eliminating effect.

ACKNOWLEDGEMENTS

We express our thankfulness to Prof. Lajos Felföldy for his generous help in all phases of our research.

REFERENCES

Balls, H., Moss, B. and Irvine, K. (1989). The loss of submerged plants with eutrophication. 1. Experimental design, water chemistry, aquatic plant and phytoplankton biomass in experiments carried out in ponds in Norfolk Broadland. *Freshwat. Biol.*, **22**, 71-87.

Bernatowicz, S. and Pieczynska, E. (1965). Organic matter production of macrophytes in the Lake Taltowisko (Mazurian Lakeland). *Ekol. Pol. A.*, **13**, 113-124.

Felföldy, L. (1974). *A biológiai vízminősítés.* VÍZDOK, Budapest.

Felföldy, L. (1976). Biological water quality. A new system for the biological qualification of water. *VITUKI Vízm. Víztechn. Kut. Eredm.*, **3**, 1-37.

Harper, D. (1992). *Eutrophication of freshwaters.* Chapman and Hall, London.

Hartman, R.T. and Brown, D.L. (1967). Changes in internal atmosphere of submerged vascular hydrophytes in relation to photosynthesis. *Ecology*, **48**, 252-258.

Johnson, W.W. and Vallentyne, J.R. (1971). Rationale, background and development of experimental lake studies in north-western Ontario. *J. Fish. Res. Bd. Canada*, **28**, 123-128.

Juday, C. (1924). The productivity of Green Lake, Wisconsin. *Verh. Internat. Verein. Limnol.*, **2**, 357-360.

Lakatos, Gy. (1976). On the photosynthesis and chlorophyll efficiency in benthic communities dominated by Vaucheria dichotoma Agh. in Lake Velencei (Hungary). *Acta Bot. Hung.*, **22**, 381-391.

Lakatos, Gy. (1978). The phenomenon and significance of benthonic eutrophication in Lake Velencei, Hungary. *Acta Biol. Debrecina*, **15**, 147-168.

Lakatos, Gy. and K. Kiss, M. (1983). Investigation of benthonic eutrophication in Hungary. *Proc. Int. Symp. Aquatic Macrophytes, Nijmegen*, 123-128.

Lakatos, Gy., K. Kiss, M., Kiss M. and Juhász, P. (1997). Application of constructed wetlands for wastewater treatment in Hungary. *Wat. Sci. Tech.*, **35** (5), 331-336.

Lorenzen, C.J. (1967). Determination of chlorophyll and pheopigments; Spectrophotometric equations. *Limnol. Oceanogr.*, **12**, 343-346.

Mason, C.F. (1996). *Biology of freshwater pollution.* Longman, Essex.

Reddy, K.R. (1983). Fate of nitrogen and phosphorus in waste-water retention reservoir containing aquatic macrophytes. *J. Environ. Qual.*, **12**, 137-141.

Schwoerbel, J. (1971). *Einführung in die Limnologie.* VEB G. Fischer Verl., Jena.

Thomas, E.A. (1968). Der Eutrophierungsvorgang bei Seen Zentraleuropes. *Wasser- und Energiewirtschaft*, **60**, 115-125.

Vollenweider, R.A. and Kerekes, J. (1980). The loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD Programme on eutrophication. *Prog. Wat. Tech.*, **12**, 5-18.

Wetzel, R.G. (1964). A comparative study of primary productivity measurements in higher aquatic plants, periphyton and phytoplankton in large shallow lake. *Int. Rev. ges. Hydrobiol.*, **49**, 1-61.

Wolverton, B.C. and McDonald, R.C. (1979). Upgrading facultative wastewater lagoons with vascular aquatic plants. *JWPCF*, **51**, 305-313.

MACROPHYTE DISSOLVED OXYGEN INTERACTIONS IN SHALLOW EUTROPHIC LAKES: A CASE STUDY (MOGAN LAKE-TURKEY)

Ayşe (Bayar) MUHAMMETOĞLU

Akdeniz University, Environmental Pollution and Control Program, 07059 Antalya, Turkey. E-Mail : bayar@ufuk.lab.akdeniz.edu.tr

Selçuk SOYUPAK

Middle East Technical University, Environmental Engineering Dept., 06531 Ankara, Turkey. E-Mail : soyupak@rorqual.cc.metu.edu.tr

ABSTRACT

Aquatic macrophytes can create undesirable consequences in relation with eutrophication processes in shallow lakes. The expected consequences are discussed by presenting the research results conducted for this purpose. A study has been initiated to investigate the interactions between dissolved oxygen (DO) and macrophyte in shallow eutrophic lakes. Mogan Lake in Ankara-Turkey has been selected for the study. The lake is classified as an eutrophic shallow lake where dense masses of macrophytes cover most of the lake especially in summer and inhibit the recreational and navigational activities. An intensive measurement and sampling program was realized for the determination of DO, temperature, macrophyte biomass, chlorophyll-a, BOD_5, COD, pH, Secchi depth, and nitrogen and phosphorus groups. Among the research findings are the occurrence of DO concentrations twice the saturated level with anoxic conditions at the bottom in summer, the occurrence of photoinhibition and the DO transport in reverse direction from the water surface to the atmosphere. Additionally, the highly seasonal and diurnal fluctuations of DO, the summer peak and the autumn decrease of DO concentrations in relation with the growth/decay of macrophytes were illustrated. The increase in the autochthonous production of organic matter was observed to lead to drastic DO depletions near the bottom layers which is an important symptom of eutrophication. As a result of macrophyte photosynthesis, high pH levels are produced which disturb life forms in the lake ecosystem. pH of the environment is a key factor in the growth of organisms and most species cannot tolerate pH levels above 9.0 or below 5.0.

KEYWORDS

Dissolved oxygen, eutrophication, macrophyte, Mogan Lake, pH, shallow lake, Turkey.

INTRODUCTION

Macrophytes are among the essential components of most freshwater ecosystems and they are physically, chemically and biologically active members of the aquatic community. Aquatic macrophytes make important contributions to the structure and dynamic processes of freshwater ecosystems (McDermid *et al.*, 1983). Their presence is of significance to man, being beneficial in some respects by providing shelter for fish and substratum for periphyton beside their aesthetic value. However, they are also detrimental especially when in profuse amounts as they impede navigation, interfere with recreational activities, impair drainage and affect the oxygen balance. Macrophytes can physically affect their environment by slowing water movement, thus retaining silt, particulate matter and other floating debris causing attenuation of light and deposition of matter. Macrophytes can chemically influence their environment by cycling oxygen, phosphorus, nitrogen, carbon, calcium, iron, sodium and many other elements and by accumulating organic matter which later causes oxygen depletion in the ecosystem. Macrophytes can biologically modify their environment by providing habitat, protection, food, and spawning substratum for other members of the freshwater community (McDermid *et al.*, 1983).

Until recently, most research on macrophytes has been limited to taxonomy, species distribution, biomass estimation and eradication techniques. This imbalance in the literature implies that ecosystem effects of macrophytes and their relation with the eutrophication process are less well known than their physiological ecology (Carpenter *et al.*, 1986; McDermid *et al.*, 1983). However, understanding the ecosystem effects of macrophytes is extremely important for the advancement of aquatic ecology. In this respect, literature on ecosystem effects of submerged macrophytes needs further studies for water quality management and better use of the fresh waters.

In this study, the ecosystem effects of macrophyte, mainly the changes in dissolved oxygen, are presented to show the extent of adverse impacts of macrophyte growth. The physicochemical conditions inside littoral submerged plant beds exhibit distinct differences from open water which affects the natural balance of the ecosystems. The strong vertical and horizontal gradients of oxygen and pH are formed because of the calming effect of the plant masses on wind-induced water movements and also the respiratory and photosynthetic activity of the plant biomass.

METHODOLOGY

Description of Study Site

Mogan Lake, located 20 km away from the city center of Ankara-Turkey, is a shallow natural lake. The mean and maximum depth of the lake is 2.20 and 4.00 m, respectively. The length of the lake is 6000 m while its mean width is 1350 m. The average water surface level is 972 m above sea level. Mogan Lake can be classified as a temperate lake due to the warm water exceeding 20°C starting from May till September, and the highest summer water temperature of about 27°C.

The catchment area of Mogan Lake is about 925 km^2. The main sources of nutrients and other pollutants to the lake is from five creeks carrying the runoff water from the watershed, and from a nearby town and its industries. Nonpoint sources which carry waters of upland farming land are also an important pollutant source (Altınbilek, 1995; Bayar *et al.*, 1997). A sediment layer rich of organics exists at the bottom of the lake. The sediment deposition rate of the lake is 236 000 m^3/yr (0.043 m/yr). Masses of emergent and submerged macrophyte cover most of the lake surface area especially in summer.

The well known OECD methodologies of eutrophication classifications could not be applied to Mogan Lake due to its shallowness and dominance of macrophytes. Different trophic level definitions have been classified for temperate lakes according to the stages of macrophyte. According to these trophic level definitions (Ryding *et al.*, 1989; Henderson-Sellers *et al.*, 1987), Mogan Lake is classified as an eutrophic lake. Eutrophic stage is observed in Mogan Lake by the increase in plant growth along with the decrease in species diversity and sensitive species. Especially the increase in macrophyte species most tolerant to pollution (potamogeton species) indicates the eutrophication process in this lake.

Measurement and Sampling Program

In order to examine the in-lake water quality of Mogan Lake, a measurement and sampling program was initiated starting from April 1996 until November 1996. Six main measurement and sampling stations were selected along the length of the lake as shown in Figure 1. In-situ measurements included the fortnight measurements of temperature, DO and secchi depth at several points along the water depth. Water samples have been collected monthly from the surface, mid-depth and bottom of the lake and analysed in the laboratory for pH, BOD_5, COD, phosphorus and nitrogen groups. Chlorophyll-a was analysed monthly just for the samples collected at the surface layer. Additional 13 supplementary stations were added for the biweekly sampling and measurement of DO, temperature and secchi depth to have a higher resolution of their distribution. Hourly measurements of DO and temperature were realized for some days. The flow rate of the creeks entering the lake and their pollution loads were measured as well. The sampling and laboratory analyses were conducted according to standard techniques (Clesceri *et al.*, 1988), while conventional field instruments were used for the in-situ measurements.

The fresh and dry weight of macrophyte biomass per unit surface area were measured biweekly at all stations (main and supplementary). A mechanical dredge sampler was used to collect macrophyte plants from a quadrat of 0.15 m^2 . The fresh weight of the whole sample was determined after draining under standard conditions for a certain time. The dry weight was determined by taking a fraction of the sample of known wet weight and re-weighing after drying it at 105°C for 24 hours. Moreover, the macrophyte species of the collected samples have been identified. The distribution of macrophyte density along the water depth and in the lake surface were inspected and registered at site during sampling.

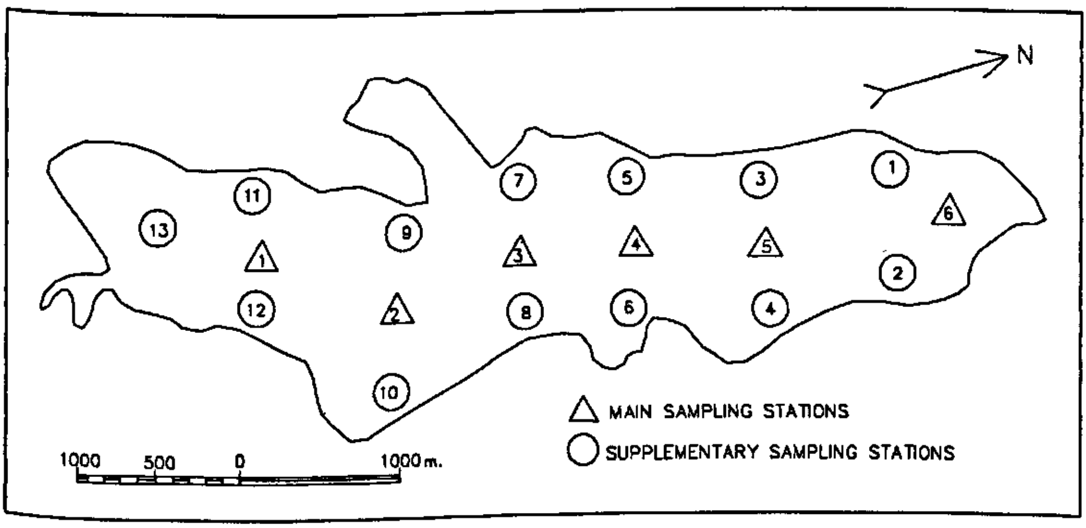


Fig. 1. Measurement and sampling stations in Mogan Lake

RESULTS AND DISCUSSION

In order to describe DO-macrophyte interactions, occurrence of supersaturated and unsaturated DO, seasonal and diurnal variation of DO, and the depth profile of DO in relation with the macrophyte biomass changes need to be investigated along with the physical and environmental conditions in the lake.

Seasonal Machrophyte Biomass and DO Variations

The density and distribution of macrophytes is influenced by the availability of nutrients and dissolved gases, light and temperature. Especially the strength and penetration of light underwater is the primary factor affecting the occurrence and productivity of macrophytes (Barbosa et al, 1989; Duarte *et al.*, 1986). Natural factors including the current, depth and nature of the stratum need to be considered. Macrophytes usually follow an annual growth cycle. They die down each winter and re-grow in spring from overwintering turions on the bottom. Machrophytes produce oxygen as a result of the photosynthesis process which leads to increase the DO to levels much higher than the saturated level. The occurrence of supersaturated dissolved oxygen concentrations is described in the study of McDermid and Naiman (1983). They reported that submerged macrophyte populations are linked to DO concentrations 230% greater than the average concentration of open water without plants.

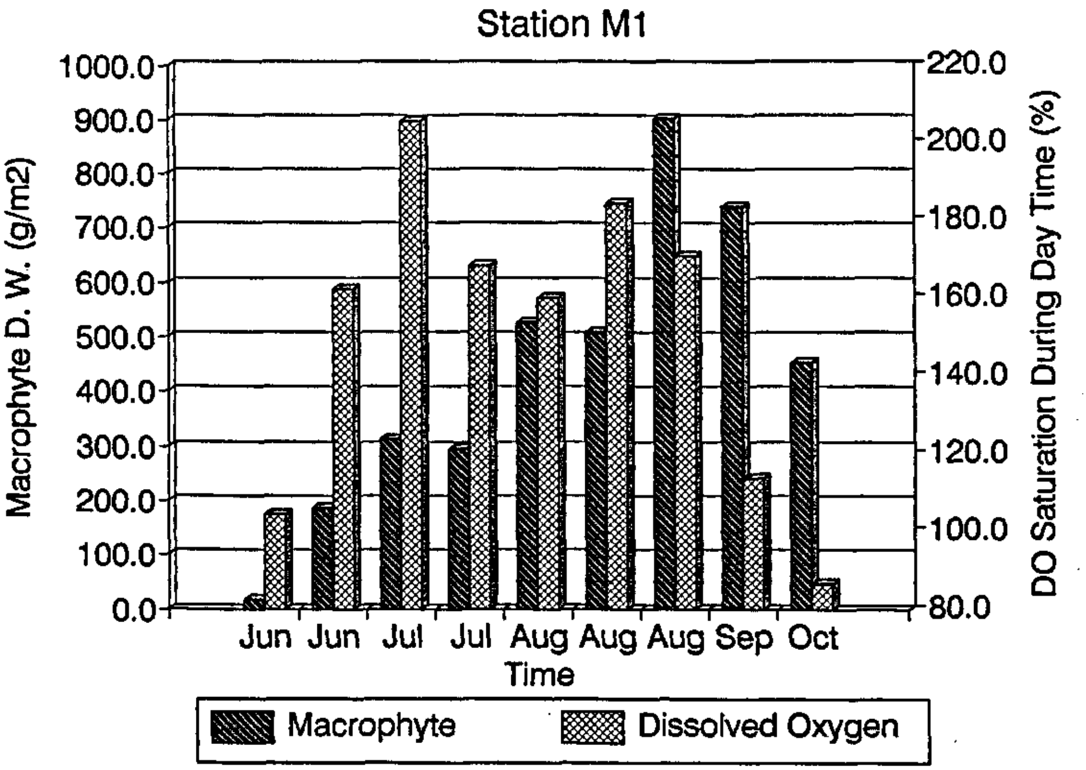


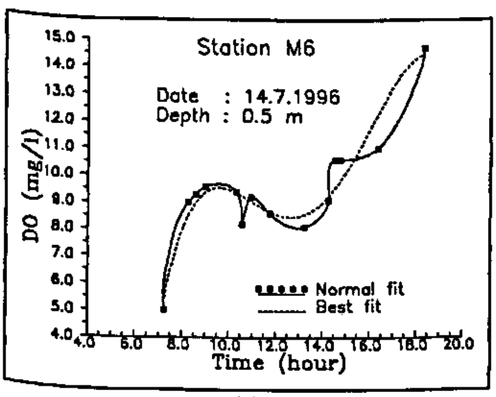
Fig. 2. Seasonal variations of macrophyte biomass and day time DO saturation (%) at station M1

In Mogan Lake, most of the lake area is covered with emergent and submerged types of macrophyte. Macrophyte biomass values are found to vary seasonally reaching its maximum levels in July and August. The DO trend measured during the day time follows nearly the same trend of machrophyte biomass. In spring, the lake exhibited DO concentrations below saturation level while in June and July DO concentrations were higher than 150% saturation at all stations and exceeded 210% saturation at some stations. This is due to the photosynthetic activity of macrophytes which produces DO. As an example, Figure 2. shows all the measurements of macrophyte dry weight (D.W.) at station "M1" together with average DO saturation percent measured in the euphotic zone. It can be noticed that the DO during the last two measurements of September and October were low. This is due to the predominance of a cloudy weather during the measurements. Seasonal changes of DO is closely affected by solar radiation, temperature and density of photosynthetic plants.

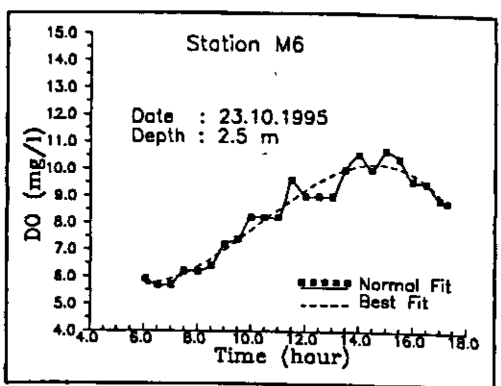
Diurnal DO Variations

While photosynthetic activity of macrophytes produces oxygen, respiration consumes oxygen. This causes considerable diurnal DO variation. Diel oxygen changes as large as 8 mg/l was observed in the waters of dense submersed macrophyte stands (Ondok *et al.*, 1984). Submersed macrophytes are recorded to oxygenate the water more effectively than floating-leaved macrophytes (Pokorny *et al.*, 1983). In another study, the diel oxygen flux in a dense macrophyte stand was observed to be twice as great as that of an adjacent harvested plot (Carpenter *et al.*, 1986). Diurnal variation of DO concentration also varies for different macrophyte species (Madsen *et al.*, 1991; Nielsen *et al.*, 1989). For example, the day and night surface layer DO saturation levels were recorded as 216% during day time and 116% at night in the stands of *Nitellopsis obtusa.*, whereas for the *potamogeton pectinatus* species they were recorded as 161% and 100%, respectively (Blindow, 1992).

In Mogan Lake, diurnal DO variation was clear in all the measurements and at all water depths along the water column. However, summer measurements showed larger diurnal changes of DO reaching up to 10 mg/l. The maximum DO concentrations were recorded in the afternoons before the sunset while the minimum DO concentrations were recorded early morning before the sun rise. Figure 3. is an example of diurnal DO variations at different depths for two different seasons in Mogan Lake. The supersaturation of DO during the day time or the reduction of DO at night may kill fish (Henderson-Sellers *et al*, 1987). The drastic diurnal DO variation can be considered among the most important adverse effects of eutrophication caused by the dense growth of macrophytes.



(a)



(b)

Fig. 3. Diurnal variation of DO at station M6
(a) at 0.5 m depth, in July
(b) at 2.5 m depth, in October

DO Variation Along Depth

Although Mogan Lake is a shallow lake with a mean depth of 2.2 m, it exhibited DO variations along the depth. In general, DO did not show much variation in the euphotic zone and the measured DO concentrations were very high (between 150-200 % saturation) during the productive season. Nevertheless, the measurements of August presented very low bottom layer DO concentrations along with the occurrence of minimum secchi depths. In August, some plankton and macrophytes start to decay and reduce the light penetration. In this respect, both sediment interactions during the death and decay of plants and planktons along with the continuing respiration process at lower layers reduce the DO concentrations noticeably by forming sediment oxygen demand. When the water temperature starts to decrease in late autumn and winter, DO variations along depth are minimized due to the decrease in reaction rates of photosynthesis, respiration and decomposition of organic matter deposited at the lake bottom.

One of the unique features of Mogan Lake is the occurrence of supersaturated DO concentrations exceeding 210% saturation levels in summer months mainly; June and July. At some stations the maximum DO concentrations were measured not at the surface but around 1.0 m below the surface due to photoinhibition effect of high solar radiation on photosynthesis rate. Another reason could be the reaeration of the lake surface which causes the loss of oxygen from the lake to the atmosphere as described by Mukhallalati (1994). Figure 4. gives an example for the occurrence of the maximum DO concentration some distance below the water surface and for the development of very low DO near the bottom. The total effect of aquatic plants on the ecosystem's oxygen balance is dependent not only on photosynthesis, but also on respiration and on plant decay processes which use up oxygen and cause drastic seasonal and diurnal variations of DO.

Seasonal pH and Secchi Depth Variations

Macrophyte metabolism can cause shifts in the inorganic carbon concentration of the water. As part of photosynthesis and carbon assimilation, machrophytes take up carbon dioxide. Therefore, the local concentration of carbon dioxide is decreased and alkalinity levels of water may increase (McDermid *et al.*, 1983).

In a lake environment, pH is normally between 6-8. When it reaches to values higher than 9, it may indicate either a photosynthetic activity or the presence of carbonates which is related to hardness of the water. In Mogan Lake, pH values started to increase by the end of June reaching its maximum during August. Most of the measured pH values exceeded 9 and even reached to 10 by mid of August. This pH trend agrees with the high photosynthetic activities during July and August. Figure 5. is an example of the seasonal pH variation in Mogan Lake.

In addition to seasonal variations of pH, high diurnal variations in the pH values may be observed when the bicarbonate ion is used as a source of cell carbon. The highly seasonal and diurnal variation of pH is not a favourable condition for fish life and algal communities. While most microalgae flourish between pH values 7 to 9, a few species can grow at very acidic or alkaline pH values. Although pH values of 5 to 9 are harmless to fish, variations within the 5-9 range may be lethal by altering the toxicity of ammonia, cyanide and other compounds (Haslam, S.M, 1990). Especially ammonia toxicity increases at alkaline pH levels (Kausch, H. *et al.*, 1994). The increase in pH values also affects nitrification process. High pH levels produced as a result of macrophyte photosysnthesis also leads to release of Fe and Al bound phosphorus which increases the in-lake nutrient concentrations.

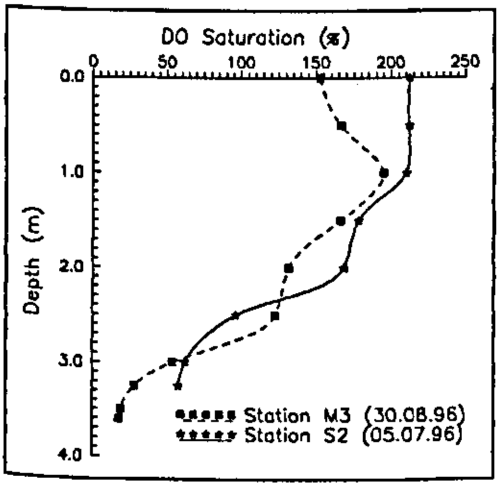


Fig. 4. Depth profile of DO

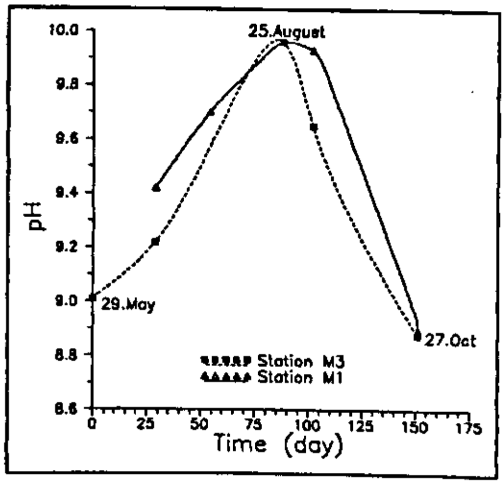


Fig. 5. Seasonal variation of pH values

With respect to secchi depth, the measurements in Mogan Lake show a seasonal variation with an average depth of 1.5-2.0 m where the water depth is about 3.5 m at most stations. Only at stations where water depth is less than 3 m, secchi depth values could reach to the lake bottom. The maximum secchi depths were measured by the start of July when the lake DO values reach the maximum of all the sampling period.

CONCLUSIONS

In this study, the interactions of macrophyte and dissolved oxygen in a shallow lake have been studied to determine the trend with respect to their seasonal and diurnal changes along with their relation with the physical and chemical properties of the ecosystem environment. Although macrophytes are considered to be essential components of most freshwater ecosystems, they can contribute significantly to the occurrence of eutrophication process. The main ecosystem effects of macrophytes in relation with the eutrophication process includes the increase in the autochthonous production of organic matter due to accumulation of dead plants, and the highly seasonal and diurnal fluctuations of DO causing the occurrence of both DO supersaturation and depletion. This situation affects the ecosytem behaviour adversely. The diurnal DO fluctuations and the depth profile of DO show the symptoms of eutrophication by drastic DO depletions during night times and near the bottom layers. Additionally, interference with the navigational & recreational activities and acceleration of the sediment deposition rate are other undesirable consequences related to dense macrophyte growth. The results of the present research will be used in the next stage for the development of a water quality simulation model which will include not only the in-lake pollutant transport and transformation processes but also the point and diffuse nutrient loadings outside the lake system. The developed model will be used for the assessment of water quality management options for different lake ecosytems.

ACKNOWLEDGEMENT

This study has been supported by the Turkish Scientific and Technical Research Council (TÜBİTAK) through YDABÇAG contract 355 and Environmetal Engineering Department of Middle East Technical University.

REFERENCES

Altınbilek, D. (1995). Gölbaşı Mogan and Eymir Lakes Water Resources and Environmental Management Planning Project, Final Report, Middle East Technical University, Ankara.

Barbosa, F.A.R., Tundisi, J.G., Henry, R. (1989). Diel variations in a shallow tropical Brazilian lake II. Primary production, photosysnthesis efficiency and chlorophyll-a content. *Arch. Hydrobiol.*, 116:4, 435-448.

Bayar, A., Soyupak, S., Altınbilek, D., Mukhallalati, L., Kutoğlu, Y., Gökçay, C.F. (1997). Use of Modelling for Development of Management Strategies in Control of Eutrophication for Lake System of Mogan and Eymir. *Fresenius Envir. Bull.*, 6: 115-120.

Blindow, I. (1992). Long and short-term dynamics of submerged macrophytes in two shallow eutrophic lakes. *Freshwater Biology*, Vol. 28, 15-27.

Carpenter, S.R., Lodge, D.M. (1986). Effects of Submersed Macrophytes on Ecosystem Processes. *Aquatic Botany*, 341-370.

Clesceri, L.S., Greenberg, A.E., and Trussell, R.R. (Editors) (1989). Standard methods for the examination of water and wastewater, 17th ed. APHA, AWWA and WPCF, Wahington, DC.

Duarte, C.M., Kalff, J., Peters, R.H. (1986). Patterns in Biomass and Cover of Aquatic Macrophytes in Lakes. *Can.J. Fish. Aquat. Sci.*, Vol. 43, 1900-1908.

Haslam, S.M. (1990) *River Pollution : An Acological Perspective*. John Wiley & Sons, Great Britan.

Henderson-Sellers, B., Markland, H.R. (1987). *Decaying Lakes*. John Wiley & Sons, Great Britan.

Kausch, H., Lampert, W. (1994). Advances in Limnology, Algae and Water Pollution. Archiv Für Hydrobiologie, H42, Stuttgart: Schweizerbart.

Madsen, T.V., Maberly, S.C. (1991). Diurnal variation in light and carbon limitation of photosynthesis by two species of submerged freshwater macrophyte with a differential ability to use bicarbonate. *Freshwater Biology*, Vol. 26, 175-187.

McDermid, K.J., Naiman, R.J. (1983). Macrophytes: Freshwater Forests of Lakes and Rivers. *The American Biology Teacher*, Vol. 45, No. 3, 144-150.

Mukhallalati, L. (1994). A Dissolved Oxygen Model For Reservoirs and Lakes With Highly Varying Inputs and Outputs, Ph.D. Diss., METU, Ankara, Turkey.

Nielsen, S.L.,Sand-Jensen, K. (1989). Regulation of photosythetic rates of submerged rooted macrophytes. *Oecologia*, 81: 364-368.

Ondok, J.P., Pokorny, J. and Kvet, J. (1984). Model of diurnal changes in oxygen, carbon, dioxide and bicarbonate concentrations in a stand of *Elodea canadensis* Michx. *Aquatic Botany*, 19: 293-305.

Pokorny, J. and Rejmankova, E. (1983). Oxygen regime in a fishpond with duckweeds (Lemnaceae) and *Ceratophyllum*. *Aquatic Botany*, 17: 125-137.

Ryding, S., Rast, W. (1989). *The Control of Eutrophication of Lakes and Reservoirs*, The Parthenon Publishing Group, Unesco.

MODELLING

CONTEMPORARY ISSUES IN WATERSHED AND WATER QUALITY MODELING FOR EUTROPHICATION CONTROL

Robert V. Thomann

Research Professor, Manhattan College, Environmental Engineering Dept., Riverdale, NY 10471

Lewis C. Linker

Modeling Coordinator, US Environmental Protection Agency, Chesapeake Bay Program, Annapolis, MD 21403

ABSTRACT

Three issues are discussed: controllability of non point nutrient loadings using watershed models; the sometimes counter intuitive results from eutrophication models from nutrient controls for coastal waters; and the potential significant interaction of improvement in habitat for suspension feeding bivalves. For the Chesapeake Bay watershed model, and for Limit of Technology (LOT) controls, a 16% and 45% reduction in nitrogen and phosphorus, respectively, is calculated. For the Bay, it is concluded tht removal of phosphorus only is less effective than nitrogen in improving bottom water DO because of differential transport of nitrogen downstream. For the Delaware estuary, a significant decline in phytoplankton chlorophyll has been observed in the absence of any nutrient controls but in the presence of improved DO. A simple model is offered tht hypothesized an increase in benthic bivalve filtration of overlying water as a result of improvement in DO.

KEYWORDS

Watershed; nitrogen; phosphorus; eutrophication; dissolved oxygen; bivalve; water quality model.

INTRODUCTION

Much progress has been made in understanding eutrophication processes and in constructing modeling frameworks useful for projecting the effectiveness of nutrient reduction strategies. Eutrophication models have proved useful in allocation of nutrient loading to achieve water quality objectives. However, for the United States, in recent years, several issues have emerged, primarily in estuarine and coastal waters that have required critical evaluation of existing models and the need to develop expanded modeling frameworks. These issues are:

(1) the degree of controllability of nutrients delivered from watersheds with significant non point sources,
(2) interaction of hydrodynamic transport, sediment fluxes and resulting water quality, and
(3) the significance of benthic suspension feeders in mediating water quality.

NUTRIENT LOADING ISSUES

Watershed Modeling: Structure, Loading and Controllability

If non point nutrient inputs to a body of water are significant relative to other sources (e.g., point sources) then it is necessary to model the nutrient fate and transport in a watershed to be able to determine the degree of controllability of the load emanating from the watershed. That is, it is necessary to estimate an "all forest" or background nutrient loading compared to the current loading to determine the amount of nutrient load that is capable of being reduced. Figure 1 below shows a compilation of nitrogen loading for the northeast US coastal region and several estuaries.

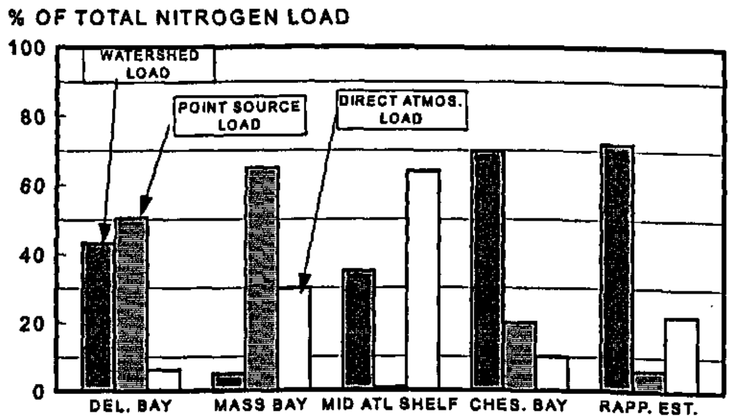


Figure 1. Relative contribution of three nitrogen inputs to various coastal waters. Del. Delaware Bay, Mass. Bay is Massachusetts Bay, Mid. Atl. Shelf is the region from Cape Cape Hatteras, Ches. Bay is the Chesapeake Bay and Rapp. Est. is the Rappahannock estuary, a small tributary of Chesapeake Bay.

As seen for several of the water bodies, the contribution from the watershed to the total nitrogen loading is significant ranging as high as 70% for Chesapeake Bay and the small estuarine tributary. In order to address management questions associated with the allocation of nitrogen loading to a water body, it is essential that the degree of controllability of the watershed loading be estimated. Also, since atmospheric nitrogen deposition occurs on the watershed, it is important to be able to calculate the contribution from atmospheric sources to the delivered load from the watershed.

Contemporary watershed models (e.g., Donigian et al, 1991) calculate the time variable response of the watershed to precipitation inputs and for varying land uses (e.g., atmospheric, agriculture, urban, forest). For most watersheds, it is generally not possible to apply classical stream quality advective-dispersive models for water quality throughout the basin because of the large number of computational segments that would be required. For example, for a drainage basin of 10,000 mi^2. approximately 100,000 stream segments would be required if an advective-dispersive model were used. This is generally infeasible. As a result, completely mixed segments are used that represent large reaches of the river drainage network. Further, since spatial integration is necessary, each sub-basin with the associated completely mixed stream reach aggregates all input loads to a single "edge of stream" load. This is illustrated in Figure 2.

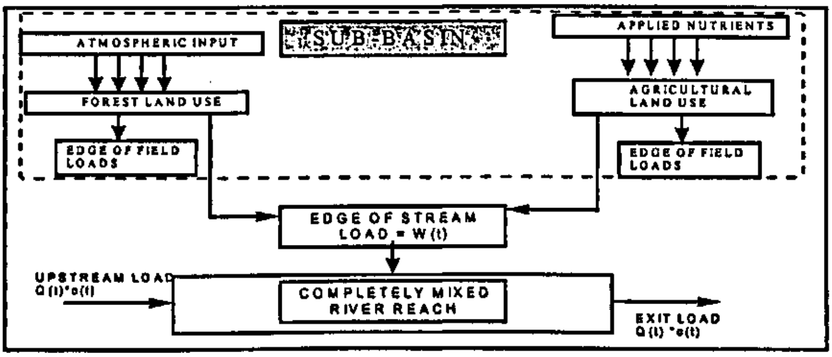


Figure 2. The principal components of contemporary large scale watershed models.

The transport of nutrients from the edge of field loading from agricultural areas through local small streams (stream order from 1-3) to the principal river is generally not modeled but integrated into a single time variable "effective" edge of stream loading. to the completely mixed river reach. As a result, the controllability of the nutrient loading is estimated on an overall sub-basin integrated level and not necessarily at the edge of the field. The watershed models therefore are calibrated to downstream water quality data in the primary river and as such represent a combination of calibration of input load and in-stream kinetics.

The Chesapeake Bay Example

The Chesapeake Bay located on the East coast of the United States has a drainage area of about 167,200 km^2 of which almost 60% is forest land (Schuyler et al, 1995). The remaining area is made up of agricultural and urban lands (about 30% and 10% of total area, respectively). Nitrogen and phosphorus inputs from the watershed together with point sources discharging directly to the Bay and atmospheric deposition on the watershed and Bay proper have resulted in two important water quality and ecosystem problems

For the open waters of the Bay, increased surface water phytoplankton biomass results in an increase in the flux of organic carbon to the sediments of the deeper waters of the Bay. Such deposition of carbon increases the oxygen demand of the sediment and decreases bottom water DO in the summer to anoxic levels (< 1 mg/L).The longitudinal summer average profile of bottom DO along the main axis of the Bay as calculated by a Bay water quality model (discussed below) begins to decline at about 100 km from the mouth of the Bay, reaches a minimum, usually less than 1 mg/L at the head end of a deep trench in the Bay which ends at about 260 km from the ocean. One of the central questions to be addressed in the management of the DO of the Bay was: What is the expected improvement in bottom water DO under various scenarios of reduction of nutrient loading to the Bay?

A modeling framework was constructed for the Chesapeake Bay system to provide a credible basis to assist the decision-making process and to further the understanding of Bay water quality processes and the sensitivity of such processes to external nutrient loading. The structure includes

1. A Watershed Model (WSM), (Donigian et al, 1991; Linker et al, 1993) to generate nutrient loads from the Bay sub-basins under (a) current conditions, (b) "all forest" conditions to provide a basis for assessing the degree of controllability of the nutrient loads, and (c) nutrient reduction scenarios using point source controls and various agricultural nutrient control practices,

2. A three-dimensional, time variable hydrodynamic model (Johnson et al, 1991a, 1991b; Dortch, 1990, Blumberg et al, 1991) to generate the transport structure of the Bay, and

3. A three-dimensional, time variable model of water quality (Cerco and Cole, 1992, 1993; Cerco, 1995) coupled to a model of sediment processes (Di Toro et al, 1992) to generate the expected water quality response of the Bay under current and historical conditions and under nutrient reduction scenarios.

The integrated latter two models herein designated as the Chesapeake Bay Water Quality Model (CBWQM) are driven by the hydrodynamic model and loadings generated by the WSM. Extensive calibration analyses of the entire modeling structure was conducted using data collected primarily during a three year period from 1984-1986 for CBWQM and 1984-1987 for the WSM.

Tables 1 is summary of the NPS nutrient loading from the watershed as calculated by the WSM and normalized to the watershed area. The range of nutrient loading from the watershed are within values reported in Thomann and Mueller, 1987 and Puckett, 1995. Of the total edge of stream loads to the various sub-basins of the watershed, it is estimated that on the average over the reference period of three years about 60% of the total nitrogen and about 40% of the total phosphorus are delivered to the Bay.

Table 1. Non-Point Source Nitrogen and Phosphorus Average Year Loadings from Chesapeake Bay Watershed Model

	Edge of Stream Loads ($kg\ ha^{-1}\ yr^{-1}$)			Load Delivered to Bay from Watershed ($kg\ ha^{-1}\ yr^{-1}$)	Ratio: Delivered Total Load to Edge of Stream Total Load
	Agricultural	Forest	Total	Total	
Total Nitrogen	14.80	3.90	10.80	6.40	0.60
Total Phosphorus	1.40	0.10	0.90	0.40	0.40
Ratio: TN/TP	11.00	39.00	12.00	16.00	

Total: Agric., Forest, Urban, Atmospheric & Point Sources on Watershed (Area=167,200 km^2)

The degree of controllability as calculated by the Bay WSM is shown in Table 2. As noted, the "All Forest" scenario is a 66% and 96% nitrogen and phosphorus reduction, respectively from a 1985 Reference case. This model output then forms a "floor" from which the controllable load can be estimated. It is also significant to note that for the Limit of Technology (LOT) scenario, a 16% and 45% reduction of total nitrogen and phosphorus, respectively from the non point watershed sources is calculated, significantly below the point source reductions.

The input TN and TP loads under LOT are about two and ten times higher, respectively, than the "All Forest" loadings. These results therefore bound the technological ability to reduce nutrient loadings from this watershed, an important management result. Watershed model runs to determine the contribution of atmospheric deposition of nitrogen indicated that about 20% of the total load is due to this source. From a management viewpoint, this indicates the potential for further reductions in TN to the Bay from control of remote nitrogen emissions to the atmosphere.

Table 2. Estimated Nutrient Loadings (normalized to surface area of Chesapeake Bay) for Different Scenarios and Percent Reduction from a 1985 Reference Case.

	"All Forest"		Limit of Technology	
	$mg\ m^{-2}\ d^{-1}$	% Reduction	$mg\ m^{-2}\ d^{-1}$	% Reduction
Total Nitrogen				
Total NPS	17.40	66.00	29.00	16.00
Point Source	0.00	-	1.70	85.00
Total	17.40	66.00	36.30	29.00
Total Phosphorus				
Total NPS	0.12	96.00	1.20	45.00
Point Source	0.00	-	0.03	96.00
Total	0.12	96.00	1.40	56.00

CONTROL OF LIMITING NUTRIENT IN ESTUARINE SYSTEMS

Estuarine and coastal systems incorporate the boundaries between fresh water, transition zones and sea water. As such, the fresh water regime may be controlled by either nitrogen or phosphorus depending on the relative contribution of point and non point sources. For the sea water end of the system, nitrogen is clearly the controlling nutrient. With time variable fresh water flow inputs, resulting varying residence times and changing contributions of watershed and direct discharges, the controlling nutrient may be difficult to assess. Therefore, an important nutrient management question for an estuarine system is: "Which nutrient should be controlled in order to achieve water quality improvements?" It is important then to assess whether nitrogen or phosphorus or both are important in controlling phytoplankton growth and subsequently in the flux of organic carbon to the sediments for subsequent oxidation. Again, the issue is to use models to determine whether there are any counter intuitive resutls to be expected. If so, the model will help to explain the occurrence of such results and therefore aid the decision process.

An example from the Chesapeake Bay described earlier is informative. The results from the CBWQM Reference case are shown in Figure 3 for the ratio of DIN/DIP.

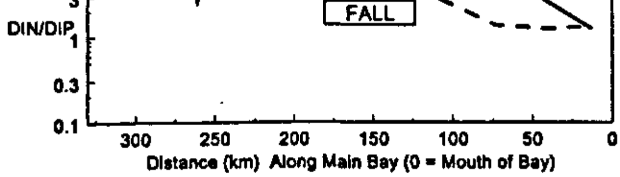


Figure 3. Calculated DIN/DIP from Chesapeake Bay Water quality Model of Cerco and Cole (1992), compiled in Thomann et al (1993)

A Redfield stoichiometry of 7 is used as a guideline for whether nitrogen or phosphorus is the controlling nutrient. For DIN/DIP > 7 - 10, phosphorus would be expected to control phytoplankton growth while for

DIN/DIP < 7 -10, nitrogen would control. As shown in Figure 3, it is estimated that during the spring, 80% of the Bay is controlled by phosphorus which reduces to about 50% of the length of the Bay by the fall. Only the lower Bay in the spring appears to be controlled by nitrogen. This calculation is in general agreement with the analyses of Bay data by Fisher et al (1992). It is concluded then that the Bay biomass is primarily controlled by phosphorus during the spring bloom period. Nitrogen is an important nutrient in the lower 20% of the Bay during this period.

The results from several nutrient reduction scenarios are shown in Figure 4 (left). The three scenarios indicate the range of response in reducing phytoplankton primary production that is to be expected from (a) Limit of Technology, (b) LOT applied to nitrogen only, and (c) LOT applied to phosphorus only. These results are consistent with what is expected. That is, the maximum reduction in primary production is from phosphorus reduction in the upper region of the Bay, while LOT-N only has relatively little effect on primary production in that region and conversely P reduction has little effect on primary produciton in the lower Bay.

While these results are consistent with known limitations, a surprising result emerged upon calculating the expected response of the dissolved oxygen resources of the Bay to reductions in nitrogen and phosphorus as shown in Figure 4 (right).

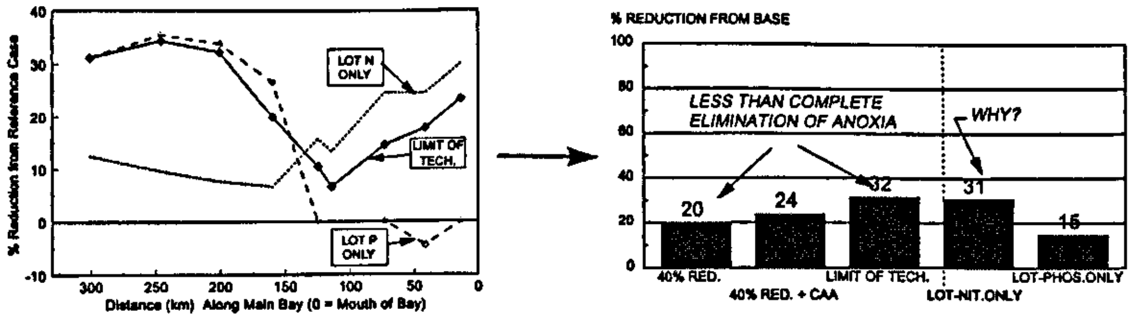


Figure 4(left). Calculated reduction in phytoplankton primary production for three control scenarios. (Right) % reduction in anoxic volume-days for 40% reduction in controllable load, 40% reduction with Clean Air Act amendments reduction in atmospheric nitrogen, LOT, LOT- N only and LOT - P only.

One owuld expect that LOT - P only would improve DO more than removing nitrogen since the lowest DO in the bottom waters is at about 260 km from the mouth of the Bay. The anoxia of the Bay is characterized by the volume and number of days tht are calculated for the Bay to be below 1.0 mg/L DO; a statistic termed the anoxic volume-days. As shown in figure 4 (right), the three scenarios representing a range from 40% reduction of controllable load to LOT improve DO resources by reducing the anoxia by 20 - 32%. But the surprising result is that LOT -P only has less of an improvement in anoxia than LOT - N only which apparently contradicts the results shown in Figure 4 (left). Figure 5 shows a possible explanation.

As shown, a detailed analysis of the model results (see Thomann et al, 1994) indicates that under P removal only, there is a calculated increase in carbon flux to the down Bay sediments which is a result of an increased down Bay transport of nitrogen. Since the lower Bay is nitrogen limited, upstream phosphorus removal results in an increase in phytoplankton carbon in the down Bay region with subsequent increase in carbon flux to the bottom. Since the bottom water net flow is from the ocean landward, the reduction from the Base case of bottom water DO is lessened. This result is illustrative of the sometimes significant interaction between nutrients and hydrodynamic transport and helps to reduce potential "surprises" under future management nutrient reduction programs.

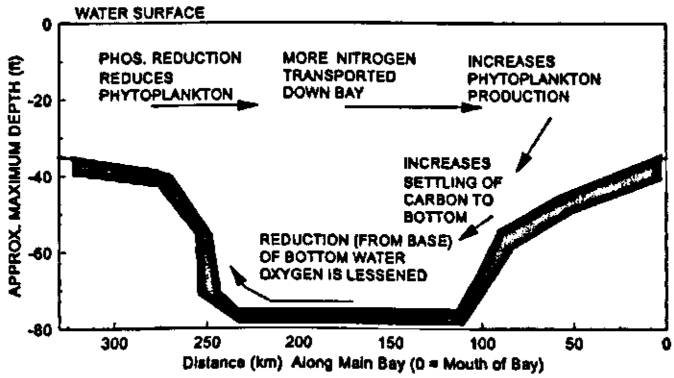


Figure 5. An explanation for the model result that removing phosphorus has less of an effect on bottom DO than removing nitrogen.

SIGNIFICANCE OF BENTHIC SUSPENSION FEEDERS

With the invasion of the zebra mussel in fresh water systems of the US, the awareness of the significant interaction of benthic suspension feeding invertebrates on water quality has increased. For the waters of the great Lakes, the impact of the mussel has been profound and coincided approximately with the reduction of external phosphorus loadings to the Lakes. A second interesting example is the Delaware estuary where for the brackish reaches of the estuary where the zebra mussel is not established, a marked increase in DO has resulted from a long term management effort to reduce carbon loading from point sources. Figure shows the increase in summer DO mean and minimum over the last three decades. Concurrently, there has been a marked reduction in phytoplankton chlorophyll with no reduction in nutrients as shown in Figure below.

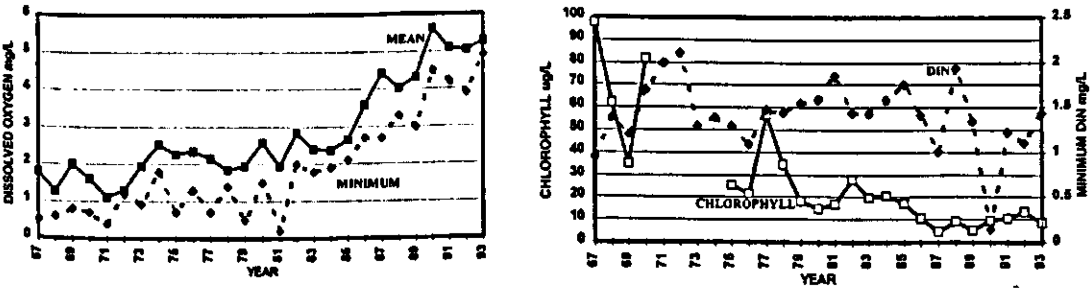


Figure 6. Water quality for mile 93 in the Delaware estuary for 1967 - 1993. (Left) Increase in DO. (Right) Decrease in chlorophyll concentration and relatively constant dissolved inorganic nitrogen. Data courtesy of Delaware River Basin Commission as compiled by Hydroqual, Inc.

A possible explanation for these results is that as the DO improved, the habitat was made more suitable for the growth of filter feeding bivalves which grazed down the phytoplankton as a function of the DO. While data on bivalve biomass is not yet available, a simple model can be used to analyze these observations. Consider a coupled pair of equations that relate phytoplankton carbon to bivalve suspension feeding biomass, as follows:

$$\frac{dP}{dt} = (G - r_p)P - (IB)\frac{P}{H} \qquad (1)$$

$$\frac{dB}{dt} = a(IB)P - r_B B \qquad (2)$$

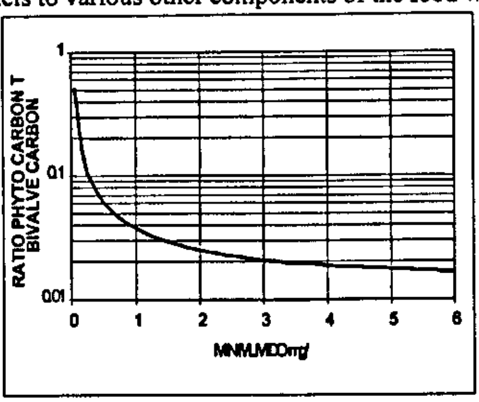
where P and B are phytoplankton carbon (mg C/m^3) and bivalve carbon (mg C/m^2), respectively, G is the growth rate of the phytoplankton (1/d), r_P and r_B represent all loss rates (1/d) for phytoplankton and bivalve carbon, respectively, I is the filtration rate of the bivalve biomass (m^3/gC-d), a is the assimialtion efficiency of the phytoplankton by the bivalves and H is the water depth. It can be hypothesized that the product IB is a function of the DO (=c), i.e., at low DO, the filtration rate and biomass are low and at high DO the product reaches some maximum value. It can also be seent aht this product is a loss rate on the phytoplankton (units of m/d). Therefore, let

$$IB = \frac{c}{c_m + c}(IB)_{max} \qquad (3)$$

Equation (2) at steady state then provides a simple relationship between phytoplankton and bivalve carbon, as follows

$$\frac{P}{B} = \left(\frac{r_B}{a(\frac{c}{c_m + c})IB_{max}} \right) \qquad (4)$$

Figure 7 below shows this relationship and Figure 9 shows the range of phytoplankton chlorophyll for nominal bivalve biomass ranging from 10-50 gC/m^2. Clearly, other combinations of parameters would give similar results. This analysis is developed simply to indicate the relationships between phytoplankton chlorophyll and potential filter feeding biomass that is DO sensitive. The significant drop in phytoplankton chlorophyll in the absence of reductions in nutrient loading on the Delaware indicates a potential important linkage between suspension feeding bivalves and chlorophyll and as a result indicates the need to expand models to various other components of the food web.



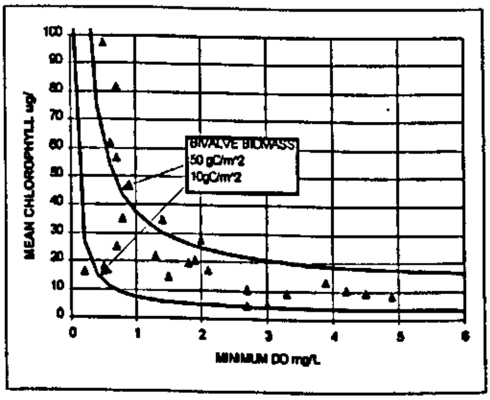


Figure 7. (Left) Plot of phytoplankton carbon (gC/m^3) to bivalve carbon (gC/m^2) using Eq. 4 with Ib_{max}=10 m/d, r_B = 0.1/d, a = 0.8, c_m=2 mg DO/L (Right) Chlorophyll vs. DO using Eq (4) with C/chloro = 50.

CONCLUSIONS

These several issues highlight the importance of having a modeling framework to assess the controllability of nutrient loadings. This is especially true as the management focus shifts from the traditional reduction in nutrients from point sources to assessing the extent of management control that can be exercised over non point sources from watersheds. For the coastal regions, the significance of hydrodynamic transport and limiting nutrient dynamics can lead to potentially counter intuitive results as was indicated by the

Chesapeake Bay example. That case indicates the importance of a credible modeling framework that incorporates state of the art processes and hydrodynamics to reduce, but not totally eliminate, future "surprises" in water quality response due to management controls. Finally, this review indicates the significant interaction that can occur between phytoplankton behavior and higher order trophic levels, specifically benthic bivalve suspension feeding animals. For the Delaware, there has been a clear decline in phytoplankton chlorophyll even though nutrient controls have not been implemented and indeed nutrient concentrations have remained well above any limiting level. The relationship between improved DO providing enhanced habitat for bivalves and subsequent increased filtration of water column phytoplankton clearly indicates a need for future models to incorporate living resource interactions in eutrophication models.

ACKNOWLEDGMENTS

Special thanks are offered to the US EPA Chesapeake Bay Program and Modeling Subcommittee for insights and watershed model results and to Dr. Carl Cerco of the US Army corps of engineers, Waterways Experiment Station, for the results of the water quality model of the Chesapeake Bay. Grateful appreciation is also given to Dr. Dominic Di Toro and Thomas Gallagher of HydroQual for their assistance and support on the Chesapeake Bay and the Delaware estuary.

REFERENCES

Blumberg, Af., B.H. Johnson, R.H. Heath, B.B. Hsieh, V.R. Pankow, K.W. Kim and H.L. Butler, 1991. Data employed in the development of a three-dimensional, time-varying numerical hydrodynamic model of Chesapeake Bay. Tec. Rept. HL-91-1, Waterways Exp. Sta., Corps of Eng., Visksburg, Miss., prepared for US army Eng. District, Baltimore, MD, 180 pp.

Cerco, C.F. and T. Cole, 1992. Application of the three-dimensional eutrophication model CE-QUAL-ICM to Chesapeake Bay. Draft Report, XIV Sections.

Cerco, C. F. and T. Cole (1993). "Three-dimensional eutrophication model of Chesapeake Bay." J. Environ. Engin. **119**(6): 1006-1025.

Cerco, C. F. (1995). "Simulation of long term trends in Chesapeake Bay Eutrophication." Journal Of Environmental Engineering **121**(4): 298-310.

Di Toro, D.M. and J.F. Fitzpatrick, 1993. Chesapeake Bay sediment flux model. Contract Rep. EL-93-2. Prepared by HydroQual, Inc. for U.S. Army Corps of Engineers, Vicksburg, Miss. 315 pp.

Donigian, A.S., Jr., B.R. Bicknell, A.S. Patwardhan, L.C. Linker, D.Y. Alegre, C-H. Chang, R. Reynolds, 1991. Watershed model application to calculate Bay nutrient loadings. Final findings and recommendations. Aqua Terra Conslutants, Comp. Sci. Corp.Chesapeake Bay Program Office, USEPA, Annapolis Md. 283 pp.

Dortch, M.S., 1990. Three-dimensional Lagrangian residual transport computed from an intratidal hydrodynamic model. Tech. Report EL-90-11,Waterways Exp.Sta., Corps of Eng., Vicksburg, Miss., prepared for US Army Eng. Dist. Baltimore, MD, 264 pp.

Fisher, T.R., E.R. Peele, J.W. Ammerman, and L.W. Harding, Jr., 1992. Nutrient limiation of phytoplankton in Chesapeake Bay. Mar. ECol. Prog. Ser. 82:51-63.

Johnson, B.H., R.E. Heath, B.B. Hsieh, K.W. Kim and H.L. Butler, 1991. User's guide for a three-dimensional numerical hydrodynamic, salinity, and temperature model of Chesapeake Bay. Tech. Rept. HL-91-20. Waterways Exp. Sta. Corps of Eng., Vicksburg, Miss., prepared for US Army Eng. District, Baltimore, MD, 41 pp + Append.

Johnson, B.H., K.W. Kim, R.E. Heath and H.L. Butler, 1991. Verification of a three-dimensional numerical hydrodynamic model of Chesapeake Bay. Tech. Rept. HL-91-7.Waterways Exp. Sta. Corps of Eng., Vicksburg, Miss., prepared for US Army Eng. District, Baltimore, MD.

Linker, L. C., G. E. Stigall, C. H. Chang and A. S. Donigian. (1993). The Chesapeake Bay watershed model. 1-9.?????????

Puckett, L. J. 1995. Identifying the major sources of nutrient water pollution. *Env. Sci. & Tech.* 29: 408-414.

Schuyler, L. R., L. C. Linker and Walters P.C. (1995). The Chesapeake Bay Story: The Science Behind The Program. *Water Science Technology* 31: 133-139.

Thomann, R. V., and J. A. Mueller. 1987. *Principles of surface water quality modeling and control.* Harper & Row, Publishers, New York.

Thomann, R. V., Collier, J. R., Butt, A., Casman, E., and Linker, L. C. (1994). "Technical Analysis of Response of Chesapeake Bay Water Quality Model to Loading Scenarios." *CBP/TRS 101/94, VIII Sec. + Append.*, US EPA Chesapeake Bay Program, Annapolis, MD.

MODELS FOR WATER QUALITY MANAGEMENT: PREDICTION UNCERTAINTY AND EDUCATED SPECULATION

Gerrit van Straten

Department of Agricultural, Environmental and Systems Technology, Systems and Control Group, Wageningen Agricultural University, Bomenweg 4, 6703 HD Wageningen, The Netherlands
E-mail: gerrit.vanstraten@user.aenf.wau.nl

ABSTRACT

This paper challenges the common belief that a calibrated and validated model is sufficient to warrant technically sound application to guide water quality planning measures. The reason is that the planning of measures intrinsically implies that the present situation is going to be changed. Consequently, the model is going to be used outside its original domain, which from a strict systems theory point of view is not allowed. Due to the complexity of water systems and their contained ecology, the structure of the developed models is almost surely not covering the changed conditions, and the prediction ability will deteriorate. Various remedies to mitigate the situation are discussed, among them over-specification combined with knowledge from similar systems, cybernetic modelling, educated speculation, and flexibility by feed-back.

KEYWORDS

Ecosystems; educated speculation; management; modelling; operating range; predictive power; reductionism; structural change; uncertainty; validity; water quality.

INTRODUCTION

This paper challenges the common belief that a calibrated and validated model is sufficient to warrant technically sound application to guide water quality planning measures. Based on a system theoretic view on models and an analysis of the special nature of water management planning problems the statement will be made that calibration is a necessary, but not sufficient condition for such applications.

The discussion in this paper is qualitative in nature, and is largely based on a series of observations and experiences over the past decades. No doubt, the examples given can be expanded easily by a careful scrutiny of the available literature (cf. Young, 1993), and the theoretical discussion can be supported by a more thorough mathematical treatment. The purpose is merely to draw attention to a fundamental problem of model application for management purposes in the water quality field.

A SYSTEMS VIEW OF MODELS

Terminology

A (mathematical) model has two major attributes: structure and parameters. The structure consists of the set of variables, and the mathematical relations between them. The parameters of the model are constants, which, ideally, should be time-invariant and should not depend upon the variables of the system. Autonomous time-variant parameters, such as $\theta = A \sin(\omega t)$, can easily be seen as a local sub-structure, with the new, truly time-invariant parameters A and ω.

A system manifests itself through interaction variables with the environment. Those that are given independently by what happens outside the system boundaries are called inputs (some manipulable: control inputs; some not under our command or not observable or observed: disturbance inputs). Those variables that are of interest to the specific application are called outputs (not necessarily all observable).

A particular useful concept of systems model is the state space representation, where x is a state vector, u the vector of inputs, y the vector of outputs, f and g vector valued functions representing the model structure, and θ the vector of constant parameters:

$$\dot{x}(t)=f(x(t),u(t),\theta)$$
$$y(t)=g(x(t),u(t),\theta)$$

but it is not necessary to take this view for the present analysis.

Another notion that is needed for the subsequent treatment is the model's 'operation range'. It is defined as the amplitude and frequency range of the model inputs over which the model output sufficiently matches the real system behaviour. More loosely speaking, it is the domain over which the model is 'valid'.

The prediction problem

The prediction problem can be illustrated as follows. Let us assume, for the sake of the argument, that we have full access to all inputs operating on the system. Suppose that there is a universal model with state variable χ, which we consider to be partitioned into χ_1 and χ_2, as follows

$$\dot{\chi}_1=\phi_1(\chi_1,\chi_2,u,\theta)$$
$$\dot{\chi}_2=\phi_2(\chi_1,\chi_2,u,\theta)$$

Now, in the very simplified case that we have modelled the system by only considering χ_1, it is easy to see that the states that have been ignored will become part of the new model's parameter vector. However, they are not constants, and even if it is possible to select the model state such that the remaining parts of the real system are not 'activated', so that the new parameters are almost constant, there is no guarantee that this property is
retained when the model is going to be used for a new input regime, or, more severely, for conditions of structural change where parts of the ϕ_2 are becoming significant. This doubtlessly over-simplified analysis also shows that very often parameters stand for sub-processes which are not taken into account. They can perhaps be called 'garbage' parameters, because they collect everything that is hidden behind it. In passing we note that the universal model does not exist: the intended purpose will always be guiding the choice of variables and processes to be included.

WATER (QUALITY) MANAGEMENT AND ITS OBJECTIVES

The principle question of water management is: how to manage (i.e. plan and operate) a water system, including its interaction with the terrestial environment, so that the water system under study can fulfill certain pre-defined functions. Examples of such functions are: source for drinking water, irrigation, drainage, waste discharge, shipping, fisheries, recreation, nature conservation.

It is important to make a distinction between two major problem domains: operation versus planning.

During *operation*, the system is 'run' by manipulating the control variables such that the system behaves in a satisfactory way. A major characteristic is that the system is run in real-time. Often, the operation range is well defined, and mathematical models that are calibrated and validated over this domain can be employed as decision support tools, or even as part of automatic control methods. Operational water quality management basically can be seen as a control problem, and, therefore, can be treated by methods from control theory. In particular, models obtained by system identification methods directly from the data (linear black-box models, neural net models) may be as good candidates for this type of application as mechanistic models. Examples of operational problems are irrigation control, storm barrier control, hydropower control. Most applications are in the field of water quantity. An example of a quality control application is the operation of surface water storage reservoirs for drinking water supply, although it is not trivial to obtain a good model in this case over the operation range.

In water quality *planning*, the situation is entirely different. The principle aim is to decide on investments, either to remedy an existing unsatisfactory situation, or to create a new system (e.g. in city development). The following situations can be observed:

a. a change of the operation range, (e.g. less pollutant load, increased flushing),
b. a change in dimensional parameters, (e.g. changing width and depth),
c. a change in the system structure, (e.g. physically: shoreline reconstruction, deflection canals, storm barriers, sediment coverage, dredging, adhesion structures; chemically: dosage of precipitants, dredging; biologically: fish management, seeding).

The challenge is to predict the system's behaviour under changed conditions. The risk is, that these predictions are wrong, because of the unexpected effects of the change itself. A model that behaves well under the present circumstances, may not under those of the future.

Before analysing the situation, three remarks are in order. The first is that the distinction between the three situations above is somewhat arbitrary. There is, in fact, a continous scale, as will become clear below. The second is that there is an equally vague region where investment decisions may become operational decisions. For instance, incidental fish catch to improve the ecological balance in shallow lakes may become into a regular fish catch measure. Likewise, dosage of chemicals can be done on a regular basis. The uncertainties in these operational situation are, at first, similar to those of the planning problem, but may, of course, become less severe as more data about the system in operation become available. The third remark is that in planning usually no high accuracy is required. Very often, the issue is to compare various alternatives, and to select the best or least risky one (e.g. Klepper et al. 1981). This can often be done on the basis of (multi-variable) ranking alone.

LIMITATIONS OF DATA-BASED MODELS FOR PLANNING

While data-based models (black box models) are often suitable for operational purposes, they are hardly of use for planning, unless the operation range is made large enough by incorporating observations made on a number of similar systems in various operating points.

A - static - example is the well-known Vollenweider approach to lake eutrophication. By collecting seasonally aggregated data on total phosphorus, chlorophyll-a and P-load for a large number of lakes, and parameterizing the empirical relationships by some correction for lake depth, a somewhat useful picture can be obtained for the overall expected effect of P-load reduction. Nevertheless this type of analysis often leads to bizarre results, in the sense that there may be no obvious physical explanation for the result obtained. For instance, the relationship

$$P_\lambda = \frac{L_s}{q_s} \frac{1}{1+\sqrt{\tau_w}}$$

where P_λ is the annual average total P concentration (mg m^{-3}), L_s the areal annual total P load (mg m^{-2} y^{-1}), q_s the hydraulic load (m y^{-1}), τ_w the filling time (y) (Vollenweider and Kerekes, 1980) has no physical interpretation, and is even dimensionally incorrect. Also, since the results are obtained for an ensemble of lakes, the statements that can be made apply just for the ensemble, strickly speaking. Quite large deviations may occur when applied for individual lakes (cf. Van Straten, 1986).

LIMITATIONS OF MECHANISTIC MODELS FOR PLANNING

Mechanistic modelling is based upon the inclusion of prior knowledge about individual sub-processes. It usually starts out from well-established conservation laws, enhanced by constitutive relations. In principle, the operation range of mechanistic models may be wider than for data-based models, due to the alledged 'universality' of the sub-model principles. The development of mechanistic models certainly has been strongly stimulated by the advent of the digital computer, and the easiness of simulation. However, as it is impossible and impractical (and even undesirable) to include every sub-process at every conceivable detail, problems are likely to occur when these models are stretched outside their development domain when used for planning purposes, as will be shown by the examples given below.

Change of operating range

What can happen with a well-calibrated model, when used outside its operation domain?

Take the extremely simple, yet illustrative, example of BOD-decay in a stretch of river downstream a sewer discharge. Already back in 1935 the famous Streeter-Phelps BOD-DO model was established to handle this situation. Just for the sake of the argument, assume that the river flow rate is constant, so that the BOD-profile can be calculated over stream-time, according to the model

$$\frac{dL}{dt} = -kL \qquad L(0) = L_0 \quad \text{at} \quad t = t$$

The initial condition applies after mixing at the waste load point. The parameter k is the celebrated BOD-decay coefficient. This model may be calibrated against data collected at various downstream points, and validated for another load and/or streamflow condition. In any particular situation, k appears to be reasonably 'constant', i.e. variations of k over stream distance are indiscernable from the profile.

Now suppose this model is used to establish the desired degree of waste water treatment in a STP at the discharge point. The assumption that the calibration value still holds for this new situation appears to be unjustified. In fact, once the new plant is in operation, the decay rate becomes considerably slower. Several reasons can be held responsible for this: the easily degradable components of the waste water, which kept k high, are taken away in the STP, leaving the river with the more difficultly degradable substances. Moreover, one may expect changes in the heterotrophic bacteria in the river and the river bed. Note that it would not have helped to enhance the model with a sediment oxygen demand process, because in the new situation the SOD is most likely going to be changed.

The change of the operating point in this example jeopardized the apparent linearity of the model by evoking sub-processes underlying the parameter. In other words, a constant parameter did not exist over the whole range of present and future operation, and the model was faulty, despite good calibration and validation results.

Change of dimensional parameters

Suppose somebody has made a thermal balance model to describe the temperature in a lake. Very good calibration results are obtained. Now, imagine one would like to use this model to evaluate the expected lake temperatures when the lake is deepened by sand excavation. The naive approach would be to just insert the new depth value in the depth parameter of the model, with the argument that the heat balance has universal value.

After the excavation, measurements reveal that the lake became stratified, a phenomenon not taken into account in the simple heat budget model. In this example, there is enough experience to prevent this abuse of a naive model. In fact, the modelling of stratification can be done with fair precision, and no skilled consultant would have overlooked this possibility.

However, the situation is less obvious if the scene is slightly changed. Suppose the question now was not just what would happen with the temperature, but also what would happen with the algae in the lake. Now, again, even with a fairly sophisticated multiple algal model calibrated for the present conditions, chances are at odds that in the new deeper lake quite different species will win the competition (e.g. Mycrocystis blooms). Here we have an example of an induced change in the structure of the (eco-)system, and its modelling in advance would have been extremely difficult.

Change of system structure

The two sections above already indicate changes in system structure induced by modifying the operating point or by introducing structural parameter changes. The phenomena appearing when the system's structure is changed on purpose are similar. Many examples could be given of unexpected changes. Consider the situation of a small receiving creek before and after the removal of a waste water discharge. As the hydraulic regime is changing drastically, so does the vegetation in the shallow shore zone. Consequently, a model describing BOD removal, ammonia removal, phosphorus and algal bloom in the original situation is not likely to do very well under the new conditions, because of the drastic changes in the system structure, and the appearance of new unexpected phenomena (cf. Grasman and van Straten, 1994).

In many such situations it is the complexity of the ecosystem that is at the basis of these surprises. There is the general principle that niches will be occupied. Therefore, the introduction of, e.g. shallow coast zones, in a lake, may have a marked effect upon the ecosystem as a whole. Even though the nich principle is easy, the modelling of it is extremely complex. The problems can perhaps be categorized into at least three groups:

a. potentially chaotic behaviour. The many non-linear interactions in an ecosystem easily lead to system equations that have, in principle, chaotic behaviour. The main characteristic is high sensitivity to initial conditions, and as a consequence, prediction of exact time trajectories becomes virtually impossible. Even though external forcing mutilate this phenomenon, it is still potentially there, and hardly understood (Doveri *et al.*, 1993),
b. the presence of spatial gradients. Local phenomena can have a marked effect upon the system as a whole (Scheffer and de Boer, 1995). This makes it necessary to model into large and sometimes impractical spatial detail. Moreover, the exact exchange between parts of the system may be difficult to establish,
c. adaptation phenomena. Even at the species level, it is well known that adaptation occurs. For example, algal growth models usually have a light use efficiency parameter, and a maximum growth rate. However, experiments show that these parameters are by no means true constants, but can be adjusted by the algae (the light use efficiency tends to go up as algae are adapted to low light, whereas the maximum growth rate goes down). This would force sub-models of considerable complication.

Even in seemingly well defined physical conditions, for instance when it comes to predict sediment transport at sea after constructing a pier or jetty, surprises often occur. Here, also, non-linear phenomena with extreme initial condition sensisitivity are probably at the basis.

A REMARK ON UNCERTAINTY ANALYSIS

Uncertainty analysis is a technique to assess as well as possible the uncertainties at prediction (cf. Beck, 1987). Uncertainties in future inputs can easily be accommodated, and are not the subject of this treaty. Parametric uncertainties can be handled if a range or probability distribution of the model parameters can be obtained from the literature. This is particularly useful for new situations on systems that did not yet exist (e.g. reservoirs). In an existing situation, the prior uncertainty distribution of the parameters may be restricted by the actually measured one by applying the Bayesian method. Drawing from the joint probability distribution by a suitable Monte Carlo method (e.g. Latin hypercube sampling to cure the curse of dimensionality) will than lead to an uncertainty distribution of the predicted output (Janssen *et al.*, 1992).

The stochastic approach is similar to an unknown-but-bounded approach due to Hornberger and Spear (1981), and further developed by others (e.g. Van Straten and Keesman, 1991). By specifying an uncertainty bound to the data, and subsequently estimating a set of parameters that give a model behaviour within the uncertainty specified, rather than a single parameter vector, an uncertainty bound can be established on prediction by Monte Carlo simulation.

It should be noted, however, that these approaches do not, in general, provide a solution to the problem of structural change. If the prior set is rich enough, the changed future may be contained in the set, but since the exact location is not known the uncertainty is unnecessarily large. If the parameter set is estimated from the data, information about changes after the measures are taken is lacking, and the uncertainty analysis can be quite misleading (this is easy to see in the river BOD decay example given above).

REMEDIES

Let us not exagerate: in many practical cases fairly good models are attainable and the problems above are of minor importance. In general, the mass and energy balance principles hold, which restrains the play-ground for ecology. It is true, however, that the interaction with the boundaries, the sediment in particular, may constitute a problem. On the other hand one can take the simple stand that accumulation of pollutants relative to the silting of the sediment is unacceptable on the long term, a view that needs no model at all to guide future water management. In practice, things are not so simple, because for instance expected restoration times after expensive measures have been taken is of political interest. Also, there is the possibility to shape nature for our needs. One could, in theory, decide to shape the North Sea as a fishing pond, and consider nutrient supply by the discharging rivers as dosage on purpose. In all these more complicated cases the use of models is called into the planning process, and hence, methods to remedy the fundamental lack of predictive power as illustrated above are of interest.

Overspecification plus cross-comparison

A seemingly straight forward approach to enhance the operating range of a model is to include more detail than would be required to describe the present situation. This is a form of so called reductionistic approach. It is based on the belief that a better model is obtained when more knowledge is included. There are two difficulties with this approach. As already pointed out by Beck (1983), the dilemma is that parameters that are needed for the parts of the model that are not excited in the present situation cannot be obtained from the available data about the system. The only way out is to collect as much information as possible about other systems with similar processes, but there are many pitfalls. For instance, it is essential that parameters about sub-processes refer to the same model formulations (e.g. there are many ways to describe temperature and light dependencies of algae species, each with its own parameter sets; handbooks like Jørgensen (1979) must therefore be used with care). The second difficulty is, of course, to decide when to stop further detailing.

In the example of the river BOD-decay, a reductionistic modelling approach would have demanded the extension of the model with a separation of the various BOD-components, a set of heterotrophic bacteria and other micro-organisms as new state variables, a description of an attachment mechanism for bacteria, an SOD compartment, including the chemical composition, etc. It is clear that this is highly impractical. In stead, as a compromise, it seems advisable to try to learn from experience in other rivers, and compose an empirical table (Thomann and Mueller, 1987) outlining k as a function of the treatment efficiency. In this way, we still have a mass-balance based mechanistic model, but supplemented with an empirical table to enhance prediction power outside the calibration domain (a form of grey-box model).

Even this simple example shows that universal models do not exist, and that striving for it is deemed to fail simply because of the obvious immensity of such a task. In addition, it is interesting to note that the fundamental question whether more detailed modelling would indeed lead to higher prediction power in the non-linear context of aquatic eco-systems, hardly has been studied and certainly has not been solved.

While being critical about models that try to encapsulate everything, this author believes that it does make sense to strive for modular model building, where the model blocks can be re-used as needed. Object oriented programming methods would facilitate sub-model re-use.

Cybernetic modelling

Quite a different possibility has not been well developed, but may well be worth while to re-visit. This is to exploit the apparent emerging overall properties of ecosystems, like maximization of biomass, maximization of survival, maximization of energy utilization or exergy. This approach is sometimes called holistic modelling or, when combined with optimization principles, cyberbetic modelling. Initial attempts in the water quality fields have been made by Straškraba (1979). Some interesting results have been obtained in other areas, like in plant breeding (describing the onset of seed formation, King and Roughgarden, 1982; Ziólko and Kozlowski, 1995), or in bio-engineering (describing the multiple food preference, Kompale *et al.*, 1986). Still having model descriptions on a fairly aggregated scale, the cybernetic approach provides a mechanism to adjust the - possibly time-variant - 'garbage' parameters so as to optimize an overall goal. One of the challenges of this approach is, of course, to find a suitable goal function that represents reality.

Educated speculation

Another option to overcome the difficulty of complex models with uncalibrated elements is to stick to the simplified models, and to speculate on the basis of qualitative knowledge about the likely direction of change of the 'garbage' parameters when the system is brought in another operating point. Van Straten and Keesman (1991) combine this 'educated speculation' with the unknown-but-bounded uncertainty method, to reproduce the changes in phosphate and chlorophyll-a in Lake Veluwe after initiating lake flushing.

In the case of the BOD example, educated speculation would entail the setting of the BOD-decay coefficient to a lower value for the reduced load condition. Assessing a range of uncertainty on the basis of expert knowledge would then give a prediction with an appropriate uncertainty band.

Flexibility by feed-back and model post-audit

Thoman (1982) makes an plead for model post-audit. Little effort in the literature has been made to re-visit the models from the eighties and confront them with the data collected later. This kind of hindcasting would be extremely helpful to increase understanding and to reveal the weaknesses of the available models.

With respect to the planning process itself, perhaps strategies which allow a step-by-step procedure accompanied by regular feed-back with model-based early warning analysis of newly incoming data may be preferable over measures that require large irreversible investments. Admittedly, this is not always possible, but it seems that there is not a systematic study of this kind of approach.

CONCLUSIONS

While in operational water (quality) management a well-calibrated and validated model is necessary and sufficient for succesful on-line operation, calibation and validation - though necessary - are not sufficient for unguided use of the model for planning decisions. This is because by the vary nature of measures planned the system's operating point, or even its structure is altered, so that the model is stretched outside its original operational domain.

Potential remedies are cross comparison, cybernetic modelling, educated speculation and feed-back, some of which deserving more intensive study than observed hitherto.

REFERENCES

Beck, M.B. (1983). Uncertainty, system identification, and the prediction of water quality. In: *Uncertainty and Forecasting of Water Quality*, M.B. Beck and G. van Straten (Eds.), Springer, Berlin.

Beck, M.B. (1987). Water quality modelling: a review of the analysis of uncertainty. *Water Resources Research*, **23** (3), 1393-1442.

Doveri, F., Scheffer, M., Rinaldi, S., Muratori, S. and Kuznetsov, Y. (1993). Seasonality and chaos in a plankton-fish model. *J. Theoretical Population Biol.*, **43**, 159-183.

Grasman, J. and van Straten, G. (Eds.)(1994). *Predictability and Nonlinear Modelling in Natural Sciences and Economics.* Kluwer Academic Publishers, 653 pp.

Hornberger, G.M. and Spear, R.C. (1981). An approach to the preliminary analysis of environmental systems. *J. Env. Management*, **12**, 7-18.

Janssen, P.H.M., Heuberger, P.S.C., Sanders, S. (1992). UNCSAM, a software package for sensitivity and uncertainty analysis. *Report 959101004.* National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Jørgensen, S.E. (1979). *Handbook of Environmental Data and Ecological Parameters*, Int. Soc. Ecol. Modelling.

King, D., Roughgarden, J. (1982). Graded allocation bewteen vegetative and reproductive growth for annual plants in growing seasons of random length. *Theoretical Population Biol.*, **22**, 1-16.

Kompala, D.S., Ramkrishna, D., Jansen, N.B., Tsao, G.T. (1986). Investigation of bacterial growth on mixed substrates: experimental evaluation of cybernetic models. *Biotechnol. Bioengng.*, **28**, 1044-1055.

Klepper, O., Scholten, H. and van de Kamer, J.P.G. (1981). Prediction uncertainty in an ecological model of the Oosterschelde Estuary. *J. Forecasting*, **10**, 191-209.

Scheffer, M., and de Boer, R.J. (1995). Implications of spatial heterogeneity for the paradox of enrichment. *Ecology*, **76** (7), 2270-2277.

Straškraba, M. (1979). Natural control mechanisms in models of aquatic ecosystems. *Ecol. Modelling*, **6**, 305-321.

Thomann, R.V. (1982). Verification of water quality models. *J. Env. Engng.*, **108** (EE5), 923-940.

Thomann, R.V., and Mueller, J.A. (1987). *Principles of surface water quality modeling and control.* Harper and Row Publ. New York, 644 pp.

van Straten, G. (1986). Identification, Uncertainty Assessment and Prediction in Lake Eutrophication, Ph.D. thesis, Technical University Twente, Febodruk Enschede, 240 pp.

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

Vollenweider, R.A. and Kerekes, J.J. (1980). The loading concept as basis for controlling eutrophication. Philosophy and preliminary results of the OECD programme on eutrophication. *Progress in Water Technology, Norway*, **12**, 5-38.

Young, P.C. (Ed.) (1993). *Concise Encyclopedia of Environmental Systems*, Pergamon Press, 769 pp.

Ziółko, M., Kozlowski, J. (1995). Some optimization models of growth in biology. *IEEE Trans. Autom. Control*, **40** (10), 1779-1783.

MODELING THE PHOSPHORUS CYCLE OF THE KIS-BALATON UPPER RESERVOIR

A. Clement[1], L. Somlyódy[1], L. Koncsos[2]

[1] *Budapest University of Technology, Műegyetem rkp.3, 1111 Budapest, Hungary*
[2] *Water Resources Research Center, VITUKI, Kvassay út 1, 1095 Budapest, Hungary*

ABSTRACT

The Kis-Balaton reservoir system, consisting of the upper and lower elements, is located near to the mouth of the Zala River. It was established for the protection of Lake Balaton against high nutrient loads. In accordance with the original plan prepared at the end of 70s the aim was that before entering the lake, nutrients - primarily phosphorus - will be removed by macrophytes. The Upper Reservoir started to operate in 1985. In contrary to plans, it became an open lake dominated by algae. Until 1991 it has removed about 50 % of the external phosphorus load yearly. However, the retention efficiency has decreased considerably after the significant drop in the external load resulting from the operation of phosphorus removal at the wastewater treatment plant of the largest town of the Zala catchment. This fact can be explained by the increasing role of the internal load. To analyze the phosphorus removal mechanisms and to understand the behavior of the sediment, different water quality models were applied. P retention is characterized by different processes acting spatially differently. Essentially, abiotic processes like settling of inorganic particulate P and the adsorption of dissolved inorganic P are responsible for the P retention in the reservoir, mostly in the neck of the Upper Reservoir. Models calibrated are felt suitable to analyze future performances of both reservoir elements.

KEY WORDS

Balaton; eutrophication; internal load; Kis-Balaton; modeling; phosphorus cycle; sediment

INTRODUCTION AND BACKGROUND

Ecology-oriented water quality models can serve as basic tools of eutrophication management. Depending on the character of the problem, many model combinations are possible, and this fact is well reflected by the large number of water quality models described in the literature (see e.g. Orlob, 1983; Straskraba-Gnauck, 1985; Somlyódy and van Straten, 1986; Thomann and Mueller, 1987). Our work did not focus on the development of a new eutrophication model, but rather on the application of state of the art tools to aid the understanding of processes in the Kis-Balaton Reservoir.

Eutrophication of Lake Balaton has accelerated by the end of the 70s. In 1976 a unique monitoring program was started on the Zala catchment to find solutions for nutrient control. It was found that about half of the nutrient load derives from the Zala watershed, polluting the Keszthely Bay. For the protection of the lake against high agricultural diffuse loads, the best solution seemed to be the establishment of the Kis-Balaton reservoir system located near the mouth of the Zala River. The original concept developed by the end of the 70s assumed that nutrients - particularly phosphorus (P) - will be taken up by macrophytes.

The Hídvégi (Upper) Reservoir (see Fig. 1) started to operate in 1985 and in contrary to the plans it became an open lake dominated by algae. Until 1991 the reservoir removed about 50 % of the external P load yearly. However, the retention efficiency has decreased considerably after the reduction of the external load resulting from the operation of P removal at the wastewater treatment plant of Zalaegerszeg, the largest town of the Zala catchment. The Fenéki (Lower) Reservoir is still under construction. A part of it, which is a reed area has been flooded at the end of 1992. Observations reveal that P retention is not satisfactory in this wetland area. These findings indicate that both reservoirs operate differently than originally believed. Thus, there is an obvious need for the revision of the existing plan (for details see Somlyódy et al., 1997a and Somlyódy, submitted) and to analyze the function of the Upper Reservoir which is the closest case study for the future Lower Reservoir.

The main goal of this paper is to study phosphorus removal mechanisms in the Upper Reservoir, with special regard to understanding the effect of the external load decrease caused by P removal in Zalaegerszeg. Results can be used for (i) estimating the future performance of the reservoir, and (ii) to calibrate a suitable model for the description of the behavior of different alternatives of the Lower Reservoir (see Somlyódy et al., 1997a). Different types of water quality models were applied for the above purposes. Since a number of approaches can work evenly, we started with data analysis and empirical models based on yearly and monthly mass balances, and continued our assessment with the usage of more complex dynamic models of differing sediment compartments.

BRIEF DESCRIPTION OF THE SYSTEM AND DATA AVAILABILITY

The surface area of the Upper Reservoir (Lake Hídvégi) is approximately 18 km^2 and the average depth is 1.1 m at operational water level (Fig. 1). The reservoir consists of two independent areas, the main reservoir and the Cassette. The latter can "trap" accidental pollutions (5 million m^3) arriving through the Zala River. Since the reservoir is shallow with a relative large surface area, and the prevailing wind direction is almost parallel with the longitudinal axes of the lake, the water is well mixed down to the bottom. The planned Lower Reservoir extends about 50 km^2, from which the Ingói-cops, a 16 km^2 reed area has already been flooded (Fig. 1).

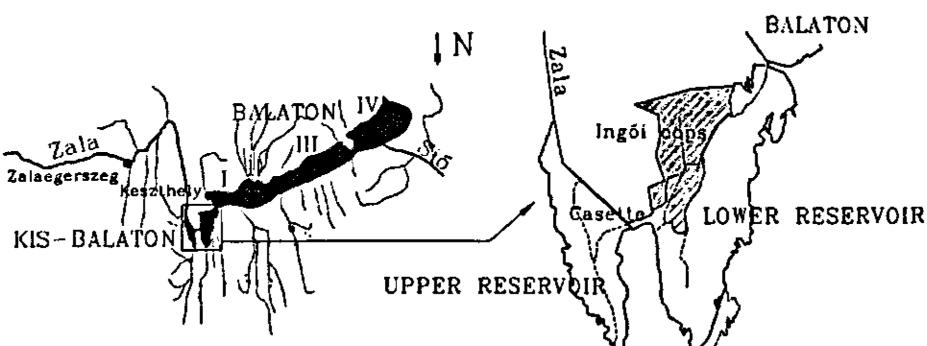


Figure 1. Lake Balaton and the Kis-Balaton Reservoir

From a water quality viewpoint, Lake Balaton and the Zala River catchment are perhaps the most "explored" areas in Hungary (Somlyódy et al., 1987). Research of the lake started at the end of the last century. Meteorological and hydrological studies date back to more than a hundred and seventy years, resp. Hydrobiological studies were launched at the twenties, while regular water chemistry observations about 30 years ago. As noted, daily flow and water quality monitoring of the Zala River was launched by the West Transdanubian Water Authorities in 1976 and was extended later to the reservoirs. These unique data (including total phosphorus, orthophosphate, total nitrogen, suspended solids, Chl-a etc.) serve as basis of our present study.

SIMPLE MODELS

Since the OECD study (Vollenweider, 1982) simple models have been used to predict effects of load reduction and to estimate trophic conditions of a planned reservoirs. The Vollenweider-type models are based on yearly total phosphorus (TP) balances of lakes under steady state conditions. Due to their simplicity they are applied frequently. The key parameter is the so-called apparent settling velocity (v_s) characterizing the loss of TP. Analyzing a number of lakes, the average value of v_s has been found to be around 10 m/year, although the variability is large. In shallow lakes the empirical relationship, $v_s = q \cdot 2 \cdot (\tau)^{1/2}$ was found, where τ is the residence or filling time and q is the hydraulic load.

The Vollenweider model was applied for annual average values of the Upper Reservoir (the residence time is about a month). TP concentrations obtained by different assumptions are displayed in Fig. 2a. As shown the model with the settling rate of $v_s = q \cdot 2 \cdot (\tau)^{1/2}$ gives an acceptable long-term average. However, calculated outflow TP concentrations exceeded measured values before 1991, and became lower than measured after 1991, when external P load decreased (primarily due to P precipitation at Zalaegerszeg). Before 1991, a v_s value of 10 m/year provides satisfactory fit to the data. Afterwards the same value results in a significantly higher P retention than the measured one. Conclusively, the apparent settling velocity changes differently than expected.

Without the Vollenweider assumption the v_s value can be estimated from the yearly TP balance:

$$v_s = \left(\frac{l_{in}}{q \cdot P} - 1\right) \cdot q \tag{2}$$

where l_{in} is the annual areal TP load and P is the annual average TP concentration in the reservoir. As illustrated in Fig. 2b, the settling velocity decreases significantly after 1991. This can be explained as the consequence of the increasing role of the likely internal load, a common feature of shallow lakes after external load reduction (Sas, 1989).

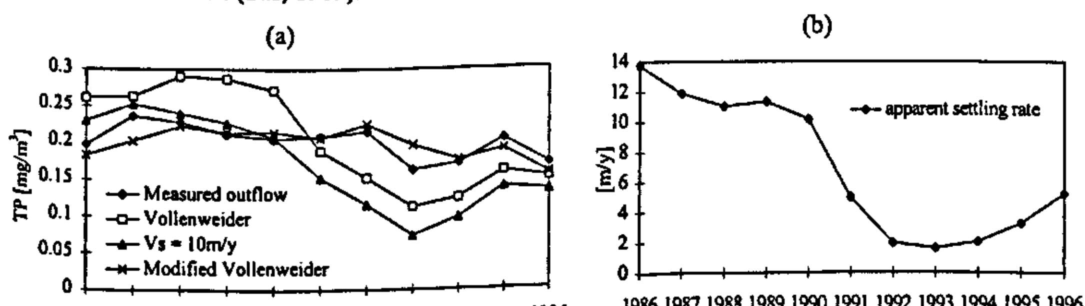


Figure 2. Application of simple models. (a) Calculated and measured TP outflow concentration. (b) Changes of the apparent settling velocity derived from TP balance

The effect caused by the P precipitation in Zalaegerszeg can be explained by the P sorption mechanism (Istvánovics and Somlyódy, 1997; Somlyódy, submitted). The internal load is attributed to high equilibrium orthophosphate concentrations maintained by the sediment deposited during the period of high specific external P load. This is high enough to saturate algal growth, and thus the biomass reaches maximum value limited by light despite the reduced external P load. Thus, the internal load is determined by the equilibrium concentrations maintained by abiotic processes and the "sudden" change in the external load resulted in a "desorption".

Simple empirical models have several shortcomings. For example, the models do not include internal P load. Taking into account it, the original formula is modified as

$$TP = \frac{(l_e + l_i)}{q} \cdot \frac{1}{1 + k \cdot \sqrt{\tau}} \tag{2}$$

where l_e is the external and l_i is the internal load. The value of the k parameter depends on individual characteristics of the system and does not necessarily equal 2.

If we suppose that the internal load was close to zero before 1991, the v_s value can be considered as the average of the period of 1986-1991. This assumption resulted in v_s=11.6 m/y, which is in good accordance with literature data (Vollenweider, 1982). Similarly, the k parameter of Eq (2) can be also estimated (k=3.7). Subsequently, the yearly internal loads can be derived from Eq (2).. The maximum internal P load reached over 30 t/y in 1993 (see later in Fig. 6a) and decreased afterwards, as expected (Sas, 1990). Conclusively, it seems justified to apply the above modified version of the simple model of Eq (2) for the Upper Reservoir (see results in Fig. 2a).

There are several other possibilities to estimate the internal load (Somlyódy et al., 1997b). For example, we can separate "winter" and "summer" data. In the latter case we assume that the internal load is zero during the winter, settling velocity is constant and equal to the winter value throughout the year. No matter, which method we have used, temporal pattern of the internal load remained unchanged: l_i was small before 1991, increased substantially afterwards, and a small decrease occurred during the last two years.

DYNAMIC MODELS

Simple mass balance models have a number of shortcomings: (i) they do not reflect seasonal or monthly changes; (ii) the role of light conditions or the photic zone do not even appear in the description; (iii) one can not distinguish long- and short-term impacts and (iv) as a consequence of the non-realistic linearity, efficiency of P removal is independent of the external loading.

Fig. 3 indicates obvious seasonal changes of the internal P load based on monthly TP balances. The seasonal pattern closely follows that of the temperature and/or the chlorophyll-a concentration.

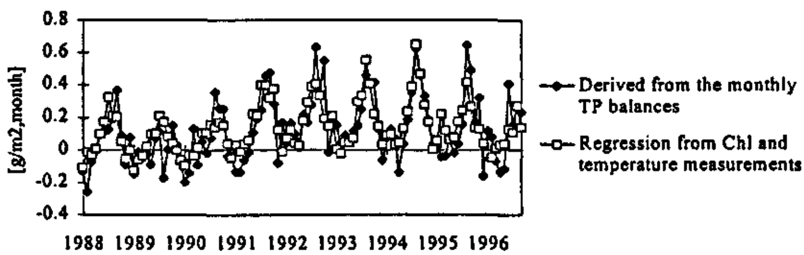


Fig. 3. Seasonal changes of internal loading in the Upper Reservoir

From this finding we can conclude, that during the growth period when much more detritus enters the sediments, the internal P load considerably exceeds the yearly average through mineralization, sorption, and diffusion. It seems therefore likely, that a well-established dynamic model may serve as an efficient tool for getting further insight on nutrient cycling and evaluating the P removal efficiency of the reservoir.

Model structures

We applied two complex model variants for the Upper Reservoir, based on the literature, careful analysis of the available data and our earlier experiences with modeling Lake Balaton (Somlyódy and van Straten, 1986; Somlyódy et al., 1997b) and other waters. At the beginning, we supposed complete mixing partially supported by dispersion calculations. Fig. 4 presents the structure of the two models.

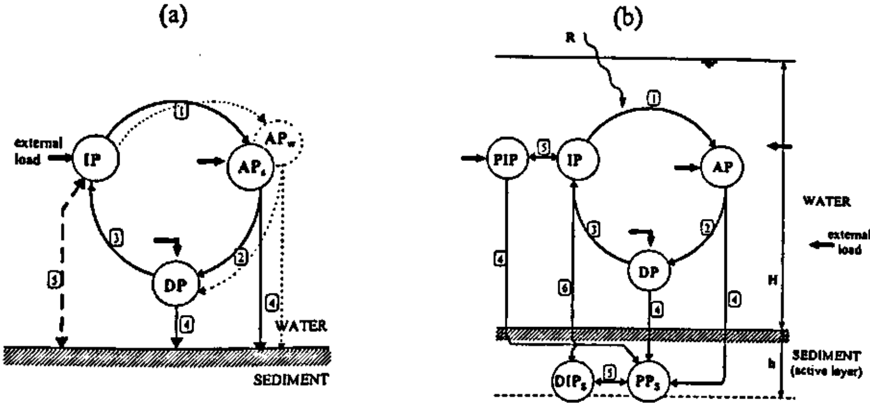


Fig. 4. Structure of the dynamic P models: (a) Model I; (b) Model II.
Notation: AP_1 and AP_2 - summer and winter algae, DP - detritus P, DRP - dissolved inorganic P, PP - particulate P, PP_s - particulate P in the sediment, DRP_s - dissolved P in the sediment; 1 - growth, 2 - decay, 3 - mineralization, 4 - sedimentation, 5 - adsorption/desorption, 6 - diffusion; R - total radiation.

Without explanation of the model equations, we briefly summarize main processes of the two models and their differences. Model I describes a simple P cycle with four state variables. The model uses "summer" and "winter" algae. Algae take up the bulk of the inorganic P that enters the water (1). Algal growth depends on P, light and temperature limitations. Background extinction in the light limitation factor is considered to be a function of the concentration of humic materials and suspended solids. We applied the same temperature function (with optimal and critical values) as in the Lake Balaton models (Kutas and Herodek, 1982; Somlyódy and van Straten, 1986). Algal decay (2) and mineralization (3) are also temperature dependent. Like for one of the Lake Balaton models, the sediment-water interaction (internal load) is a simple function of the "equilibrium concentration" that aggregates sorption and diffusion (5) of both inorganic particulate P and sediments (Somlyódy and van Straten, 1986). In the concentration range above the equilibrium a P adsorption, whereas below this range a P desorption takes place.

In contrast Model II contains only one algae variable and a detailed sediment submodel, as shown in Fig. 4. The interaction between DRP and PP is described by an equilibrium concentration both in the water and the sediments (5). Particulate P enters the sediment via settling (4), and recycling (6) is possible due to diffusion that depends on sorption and several other processes. The submodel includes a P sorption isotherm that describes the relationship between the sediment P pool and the dissolved inorganic P concentration of the interstitial water. This isotherm supplies Model II with a "memory", and thus it may suitably describe feed-back effects, following changes of loads, as well as to estimate long-term behavior of the system (for the concept see Lijklema et al., 1986). The active sediment layer is a function of the SS retention, carbonate removal and burial.

Calibration Results

Input variables are water flow, P load data, water temperature, and solar radiation. In most cases, daily data were available for the period of 1988-1996 (otherwise interpolation was employed). Different P fractions from the external load enter P compartments according to detailed studies on P fractions (Istvánovics and Somlyódy, 1997). Models were calibrated and validated by using TP, DRP, and Chl-a data.

Certain years were chosen for calibration, and independent years for validation. As an example, Fig. 5 shows the results of Model I: the calibration for a single year and the dynamic behavior for the full time period. There is an obviously good agreement between measured and calculated values. However, the model does not "exactly" reflect summer biomass peaks, or sudden short-term changes, an behavior well-known from the literature (a year-to-year adjustment of a few key parameters would result in a "better" description).

Results obtained by Model II were essentially similar, but due to the diffusion it could better fit the peak concentrations.

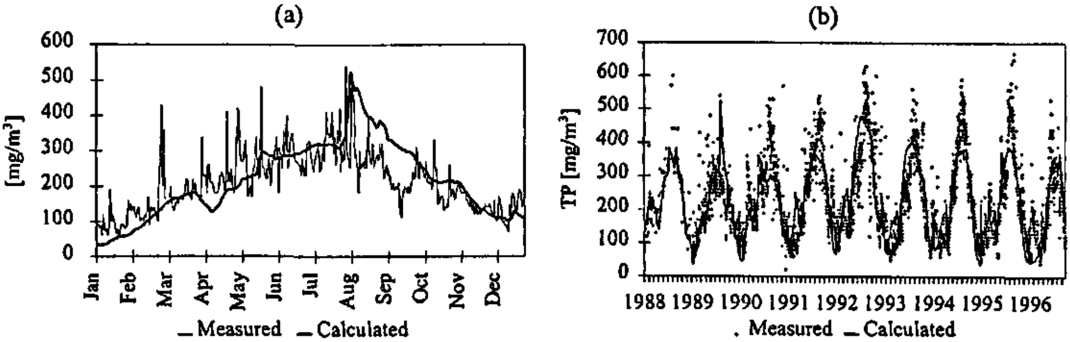


Figure 5. Application of Model I to the Upper Reservoir: (a) calibration and (b) validation

The parameters obtained were in good agreement with literature for both models. However Model I did not provide satisfactory agreement with measured data if parameters were kept constant and the exchange rate of the simplified "sorption" equation had to be adjusted. Reasons are twofold: (i) the sediment model is too simple; (ii) the fully mixed reactor assumption does not allow the description of longitudinal changes, which causes an inaccuracy in calculating the internal P load. The latter can be eliminated by using a series of reactors, instead of single one.

The parameters of Model II were constant, but the calibration with Monte-Carlo simulation resulted in a rather special isotherm (for details see Somlyódy, submitted). This "Koble-Corrigan" equation used is a generalized version of the Langmuir isotherm. According to the calibration, the initial increase of the sediment P concentration as a function of the interstitial DRP concentration is slight, but suddenly a "front"-like change occurs. This sudden rise is followed by a slow increase that approaches the saturation level. Using the Koble-Corrigan isotherm, we could satisfactorily describe both the dynamic behavior of the Upper Reservoir and the sudden response to the external load reduction, however the feasibility of the isotherm is questionable. Eventually, also here the introduction of at least two spatial segments led to satisfactory results (Somlyódy, submitted).

The internal P load was estimated with both models. Model I gave results similar to the modified Vollenweider formula with the v_s=const assumption (Fig. 6a). It is obvious since we set the settling of particulate P fractions equal to the v_s developed earlier. The second model - due to the rather detailed sediment submodel - resulted in seemingly different results (Fig. 6b).

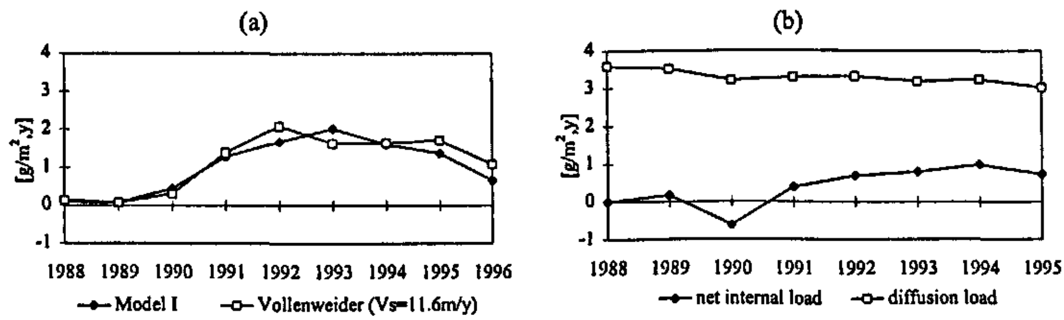


Figure 6. Estimation of the internal P load: (a) Modified Vollenweider model with v_s=11.6 m/y and Model I.; (b) Diffusion load and the "net internal load" resulted from Model II.

The diffusion load is almost constant during the full period and about the double of the maximum values resulted by Model I. The figure also shows the "net internal P load" in which the sorption and diffusion processes including the PIP settling were aggregated. The latest is comparable with results of Model I.

Two-box model

To understand driving mechanisms of the internal load and to analyze the role of processes in space a "two-box" model was developed for the Upper Reservoir. The structure of this model is similar to Model I, but the reservoir is divided into two fully mixed tanks in series. The first, smaller tank represents the upper part of the reservoir near to the Zala inflow, while the second one represents the rest of the reservoir. In contrast to the "one box" model the calibration resulted in a constant parameter set (also for Model II.). Fig. 7a shows the comparison of measured and calculated P retention efficiencies (for both models) and the overall good agreement.

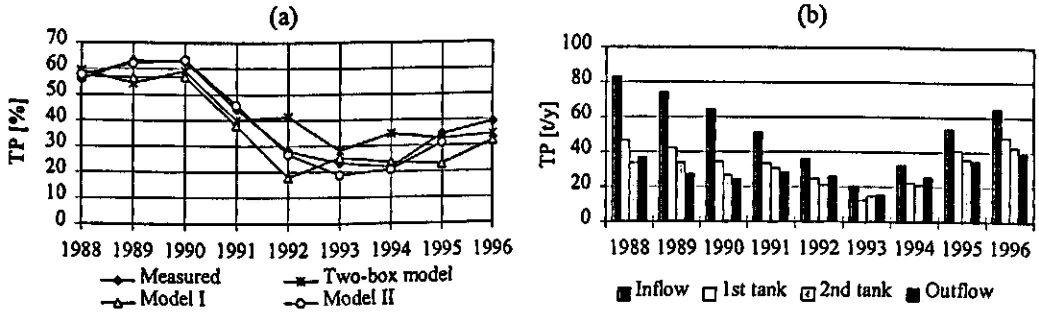


Figure 7. Summary of results gained with dynamic models: (a) Calculated and measured P retention; (b) Outflow TP fluxes of the two-box model compared to the inflow and outflow measurements.

Fig. 7b describes well what happens in the "neck" and in other parts of the reservoir. The external load is mostly removed in the upper part, due to the rapid settling of inorganic particles and the adsorption of dissolved inorganic fractions of high concentration before 1991. There is no essential difference in P retention between the two representative periods (before and after 1991). It is noted that the "two-box version" of Model II gave similar results, with the new sorption isoterm (Somlyódy, submitted).

DISCUSSION ON THE INTERNAL LOAD

The calibration of the two-box model allowed to give a more detailed explanation of the internal load. As can be seen in Fig. 8a, different processes take place in the different segments of the reservoir: the "internal load" is negative in the first reactor and almost constant and positive in the second one.

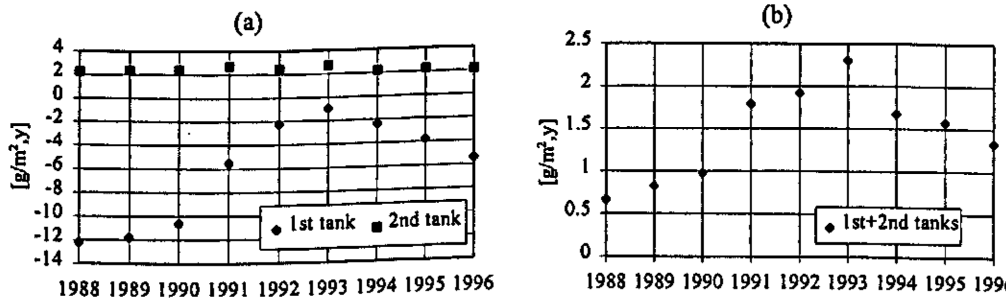


Figure 8. Internal load derived from the two-box model: (a) Separation of tanks; (b) Aggregated internal load

The "neck" of the reservoir is basically characterized by permanent adsorption. There is a considerable decrease in the adsorption after the external load reduction which reaches its minimum in 1993. A slow increase can be observed subsequently: the equilibrium P concentration decreases with the deposition of fresh sediment (Istvánovics and Somlyódy, 1997). In the second tank the internal load is strongly coupled with algal P assimilation, which decreases the orthophosphate concentration below the equilibrium. Consequently, the seasonal dynamics of desorption follows the algae growth, in agreement with Fig. 3 (Somlyódy et al 1997a).

CONCLUSIONS

Different P cycle models were applied to the Upper Reservoir to analyze the phosphorus removal mechanism with special regard to understand the behavior of the sediment. The results show that the reservoir is characterized by spatial differences in the role of different processes. The small upper part ("neck") has an important role in removing P due to the settling of inorganic particulate P and the adsorption of the dissolved inorganic fraction. Essentially these abiotic processes are responsible for the P retention in the reservoir. Yearly balances indicated that the internal P load increased after the reduction of the external load in 1991. In fact, not the internal load increased but due to the drop of the inflow orthophosphate concentration (close to the equilibrium) adsorption decreased. The largest part of the reservoir is characterized by P desorption coupled to algal P assimilation during the full operation period.

Concerning the behavior of the sediment, we can suppose a slow decrease in the equilibrium P concentration in the future (as the new sediment is deposited), resulting in a decrease in algal biomass. P retention efficiency of the reservoir would increase in proportion to the biomass decrease. The two-box version of the Model II due to its "memory" is suitable to describe the future performance of the Upper Reservoir. Since there would not be significant differences in the functioning of the Lower Reservoir in comparison to the largest part of the Upper Reservoir, Model II can be reasonably used for prediction purposes.

ACKNOWLEDGMENTS

The research was financially supported by VIZITERV Ltd. We wish to thank the staff of the West Transdanubian Water Authority for their cooperation and particularly to P. Pomogyi for her many-sided, valuable assistance.

REFERENCES

Istvánovics, V. and Somlyódy, L. (1997): : Functioning of the Upper Kis-Balaton Reservoir. (in Hunharian), Budapest University of Technology, *Manuscript*, Budapest, p. 62

Kutas, T. and Herodek,S. (1982): A Short Description of Balaton Eutriphication Model (BEM). SZTAKI Working Paper MS/9.

Lijklema L., Gelencsér P., Szilágyi F. and Somlyódy,L. (1986): Sediment and its Interaction with Water, pp. 115-182. *In: Somlyódy L., and van Straten G., eds. (1986): Modeling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton.* Springer Verlag, Berlin, p. 386

Orlob, G. T. (Ed.) (1983): Mathematical Modeling of Water Quality: Streams, Lakes and Reservoirs. *Wiley- IIASA International Series on Applied System Analysis, Vol. 12,* Chichester, UK: Wiley

Sas, H (1990): Lake restoration by Reduction of Nutrient Loading: Expectations, Experiences, Extrapolations. *Verh. Internat. Verein Limnol.* 24: p.247-251.

Somlyódy, L. and van Straten, G.(1986): Modeling and Managing Shallow Lake Eutrophication, With application to Lake Balaton. *Springer-Verlag*, Berlin, p. 386

Somlyódy, L., Herodek, S., Aradi, Cs., Clement, A., Istvánovics, V., Koncsos, L., Molnár, E., Rátky, I., Simonffy, Z., Szilágyi, F. and Várallyay, Gy. (1997a): Revision of the design of the Kis-Balaton Lower Reservoir. Synthesis Report (in Hunharian), Budapest University of Technology, Manuscript. Budapest, p. 152

Somlyódy, L., Komcsos, L. and Clement, A. (1997b): Application of phosphorus cycle models on the Upper Kis-Balaton Reservoir. (in Hungarian), Budapest University of Technology, *Manuscript*, Budapest, p. 50

Somlyódy, L.(submitted): Use of models in eutrophication management: the Kis-Balaton Case. *Submitted to Specialist Symposium and farewell reception for Lambertus Lijklema*, 28-29 August, Wageningen, The Netherlands

Straskraba, M. and Gnauck, A.H. (1985): Freshwater Ecosystems. Modelling and Simulation. *Developments in Enviromental Modelling 8.* Elsevier

Thomann,R.V. and Mueller,J.A.(1987): Principles of Surface Water Quality Modeling and Control *Harper & Row, Publishers*, New York, p. 644

Vollenweider, R. A. and Kerekes, J. J.(1982): Background and Summary Results of the OECD Cooperative Programme on Eutrophication. *OECD Report*, Paris

EUTROPHICATION VULNERABILITY ANALYSIS IN SURFACE WATERS

José M. P. Vieira[*], José L. S. Pinho[*] and António A. L. S. Duarte[*]
[*] *Department of Civil Engineering, University of Minho, 4709 Braga Codex, Portugal*

ABSTRACT

River Cavado water quality variability was studied for eutrophication vulnerability assessment at a new surface water supply intake. Since the river flow regime is artificially controlled by upstream multipurpose reservoirs, mathematical modelling was applied in evaluating alternative management scenarios. Due to the fact that surface water quality at intake location is mainly affected by a 5 km upstream wastewater treatment plant effluent discharge, algae and nutrients concentration simulations have been worked out in order to identify critical situations. Different algal concentration profiles along the river were obtained for local conditions of light energy, water temperature and estimated nutrient loads, showing high probability of eutrophication occurrence for some of the simulated scenarios. Results discussion of this study appears to be very useful for river basin wide water management policies evaluation.

KEYWORDS

Algal growth; eutrophication modelling; limitative factors; sensitivity analysis; water quality assessment.

INTRODUCTION

Problems related to water supply uses can range from shortage of water to water quality degradation mainly due to intensive urbanisation and industrial policies as well as uncontrolled agricultural practices. The presence of macroscopic plants causing blockage of intake screens, and microscopic planktonic algae producing taste and odour, must be investigated and controlled in order to protect the quality of river water intake for water supply purposes.

The river Cavado basin (Fig. 1), located in the north-western region of Portugal, has a very intensive use for water supply, agricultural irrigation and hydropower generation. New water supply project serving the Oporto metropolitan area, with a population of 0.9 million people and a design flow of 2.7 m^3/s, will introduce new challenges in the river water quality management.

Eutrophication vulnerability analysis of river water, in particular at the intake location of planned water treatment plant (Areias de Vilar WTP), was carried out in order to predict algae content in raw water. A comprehensive water quality data collection programme was followed for characterisation of present situation in the river basin. Data analysis and different scenario alternatives for the river basin management were simulated using mathematical modelling. Critical detention times and hydrometeorological conditions for eutrophication occurrence at the intake location were worked out in order to define operation conditions for upstream dams.

STUDY AREA

With a drainage area of 1589 km^2 and a mean width of 16 km, river Cavado basin has a mean elevation of 564 m with several peaks above 1500 m, and an average population density of 200 inhabitants/km^2 (minimum of 22 at Montalegre and maximum of 1770 at Braga). The annual mean rainfall is 2200 mm, 42 % of which is concentrated in the months of December, January and February. The water is intensively used for hydropower generation, domestic and industrial water supply and agricultural irrigation. Main tributaries are river Rabagão (left side, with a drainage area of 257 km^2) and river Homem (right side, with a drainage area of 246 km^2). Due to the river basin characteristics, six large hydropower plants (apart from several other small units) are in operation with an installed power of 377.6 MW and a mean annual energy production of 1535 Gwh. A total reservoir volume of 1170 hm^3 is possible with these dams, which represents a high regulatory capacity for river flows. For this reason, this water use constitutes a very important factor to be considered in any water management policy adopted for the basin.

The study area occupies the lower level part of the basin, where the main residential and industrial areas are located, and distributed for five municipalities: Amares, Vila Verde, Braga, Barcelos and Esposende. All of these municipalities are served by WTP and wastewater treatment plants (WWTP). For modelling purposes, the river reach considered in this study begins downstream Caniçada dam (Ponte do Porto) and ends near the river mouth, in a distance of approximately 48 km; the flows of River Homem are considered as a point discharge downstream Braga WTP; wastewater inputs from the urban areas include Amares and Braga WWTP effluents and untreated domestic and industrial discharges from the municipality of Barcelos.

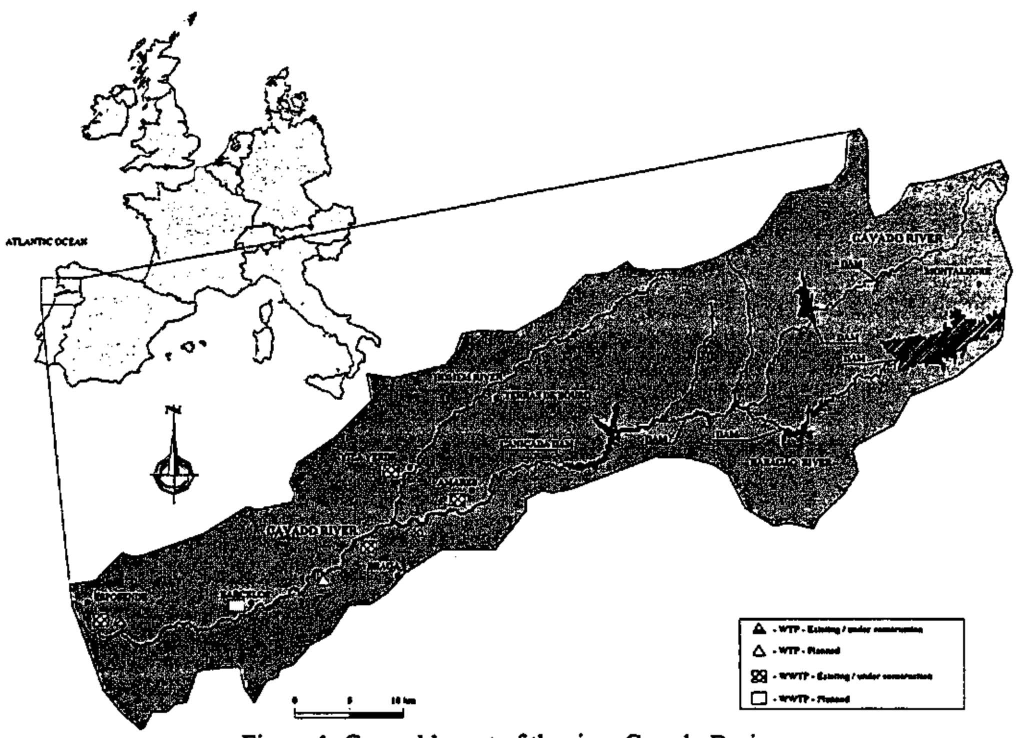


Figure 1. General layout of the river Cavado Basin.

MODEL DESCRIPTION

DUFLOW model (ICIM, 1992) was designed to cover a large range of applications, such as propagation of tidal waves in estuaries, flood waves in rivers, operation of irrigation and drainage systems and water quality problems. The package is based on the one-dimensional partial differential equation that describes non-stationary flow in open channels. As the relationship between quality and flow gets special attention nowadays and this package is suitable for modelling both, it becomes a useful tool in water quality management. In the water quality part the process descriptions can be supplied by the user. In this study, an eutrophication model based on EUTRO4 (USEPA, 1992) has been applied. The basic transport equations used in DUFLOW, which are the mathematical translation of the laws of conservation of mass and of momentum, are discretized in space and time using *the four point implicit Preissmann scheme*. This scheme is unconditionally stable, shows little numerical dispersion and allows non-equidistant grids. It computes discharges and elevations at the same point.

The quality part of the DUFLOW package is based upon the one dimensional transport equation. This partial differential equation describes the concentration of a constituent in a one dimensional system as function of time and place. The production term of the equation includes all physical, chemical and biological processes to which a specific constituent is subject to. The eutrophication model (EUTROF1) includes nitrogen, phosphorus and oxygen (Fig 2). The growth of one phytoplankton species is also simulated. The interaction between the sediment and the overlaying water column is not included in a dynamic way. However, sediment exchange fluxes of oxygen, ammonia and phosphorus may be specific location and time dependence (reflecting temporal and seasonal variations), that can be specified by the user.

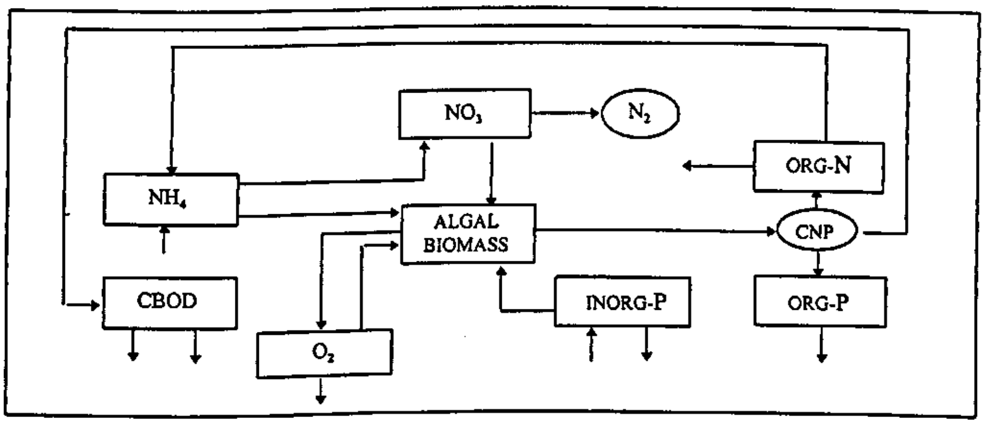


Figure 2. EUTROF1 state variable interactions (ICIM, 1992).

The model is suitable to study the short term behaviour of systems, like to examine the impacts of a discharge on the oxygen dynamics or to explore the effects of flushing on the *chlorophyll-a* concentration. Algal growth is considered to be limited by nutrients, light and temperature. It is assumed that algae can use ammonia and nitrate for their growth. The uptake of both nitrogen constituents is controlled by the ammonia preference factor. Assumptions and processes can be found in the model user's manual (ICIM, 1992).

METHODS

Data Analysis

River flows were obtained from continuous water discharges measurements in Caniçada dam (DPH/PHGP,1996). The flow rates are significantly influenced by upstream hydropower plants operation, which have an impounding effect of 40 % of the basin mean annual runoff. The reservoirs regulatory capacity effect allows monthly low-flow augmentation in dry-weather conditions. For short operational periods, weekly and daily fluctuations must be considered in the river flow regime.

Water quality data used for simulation purposes were obtained from intensive field sampling programmes (PGHIRN, 1992, and University of Minho, 1995). Model segmentation assumes thirty different river reaches each of them including or a tributary input, or a wastewater discharge, or an abstraction water point. Since weirs have been noticed to play an important role in the river flow regime, nine main weirs were considered as shown in Fig. 3. Nutrients and chlorophyll-a concentrations at Ponte do Porto were adopted as water quality boundary conditions and as initial conditions for all river sections. Braga WWTP discharges, without treatment (simulating failure risk) was considered as unique nutrient point source (Table 1).

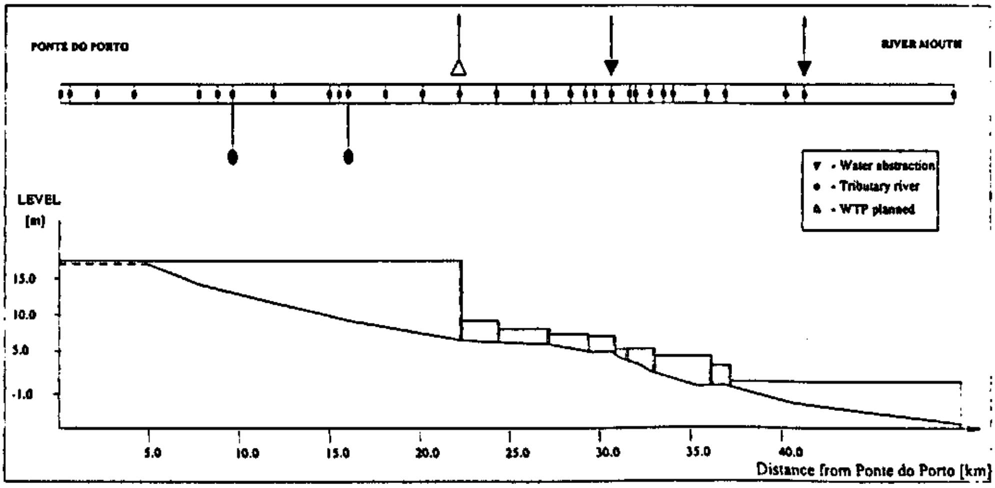


Figure 3. Model schematization and longitudinal profile.

Non-point source loads from agricultural and forest areas were estimated using literature values (Thomann, 1987). The results obtained for the most severe values are negligible when compared with those of Braga WWTP discharge.

Table 1. Average concentration of nutrients and chlorophyll-a.

Location	Phosphorus (mg P/L)	Ammonia (mg N/L)	Nitrate (mg N/L)	Chlorophyll-a (µg/L)
Ponte do Porto	0.07	0.06	0.86	3.8
Braga WWTP	9.90	31.82	2.70	---

Model Calibration

Calibration procedure consisted of comparing actual measured data values with simulated model output results. The river hydrodynamic behaviour was approximated with weir water levels error of ± 2 cm. Dissolved oxygen analysis was performed calibrating the model for the most relevant parameters (Vieira et al., 1996). Eutrophication model was calibrated for chlorophyll-a using data obtained from field survey in six sampling points (Fig. 4). Adopted model parameters fall in common default ranges stated in similar studies (ICIM,1992) and a chlorophyll-a to carbon ratio of 30 µg Chl-a/mg C was considered (Table 2).

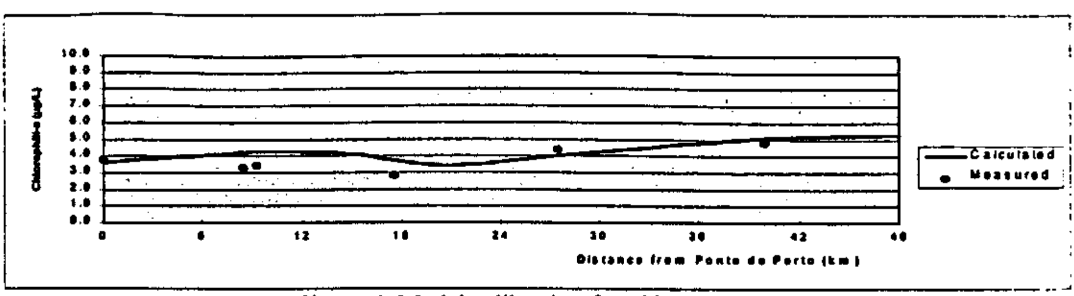


Figure 4. Model calibration for *chlorophyll-a*.

Table 2. Adopted model parameters.

Parameters	Value	Dimension
Background extinction	1.0	m^{-1}
Optimal light energy	100	$W.m^{-2}$
Monod constant nitrogen	0.01	mg N/L
Monod constant phosphorus	0.005	mg P/L
Respiration rate constant	0.1	d^{-1}
Die rate constant	0.2	d^{-1}
Maximum specific growth rate algae	1.5	d^{-1}

Sensitivity Analysis

A sensitivity analysis was carried out to evaluate the influence of model parameters with large range of variability: optimal light energy; maximum specific growth rate algae; background extinction; die rate constant; respiration rate constant; Monod constant for nitrogen and phosphorus. Significant results for two different WTP abstraction points (Areias de Vilar and Esposende) are depicted in Fig. 5, where the relative increase of chlorophyll-a concentration is shown for the most influent parameters under simulated frequent conditions (Ponte do Porto flow: $45m^3/s$; water temperature: 15 °C; light energy: 150 $W.m^{-2}$).

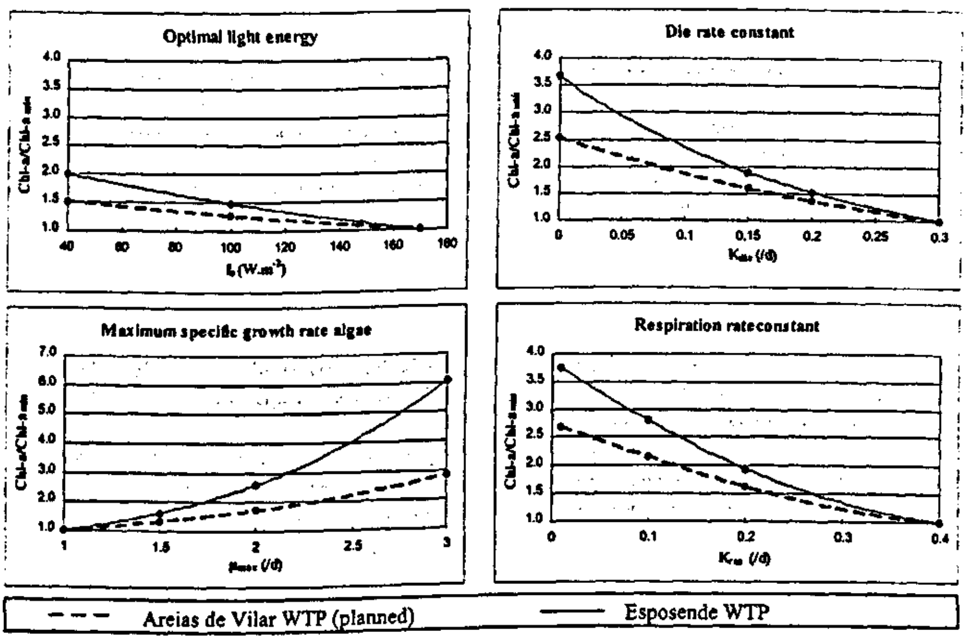


Figure 5. Model parameters sensitivity analysis.

RESULTS AND DISCUSSION

Simulated Scenarios

After model calibration, various receiving waters alternative conditions were considered. The scenarios worked out are summarised in Table 3, where variation of flow discharged at Ponte do Porto and algae growth limitation factors (light energy, water temperature) are considered. Adopted mean daily light energy values – 150 and 300 $W.m^{-2}$ – represent typical frequent and exceptional summer situations, respectively. For water temperature, two values were considered: 15 °C and 20 °C, corresponding to mean and maximum mean of observed values, respectively.

Table 3. Simulated scenarios.

Light Energy (W.m⁻²)	Water Temperature (°C)	Ponte do Porto discharges [m^3/s]		
		8	16	45
150	15	S1	S5	S9
	20	S2	S6	S10
300	15	S3	S7	S11
	20	S4	S8	S12

These scenarios represent the most relevant situations for evaluation of the river water trophic conditions, considering algal and nutrients concentration trends and DO profiles in the river reaches. The impact of Caniçada dam discharge flow can be evaluated in frequent (S1, S5, S9) and exceptional (S4, S8, S12) light energy and water temperature conditions. Comparison of scenarios S1-S3, S5-S7, S9-S11, allows the evaluation of the light energy effect in the river system for typical values of discharged flows and water temperature. The water temperature effect under extreme light energy conditions for different flows can be analysed by means of scenarios S3-S4, S7-S8, S11-S12. For this short term analysis, different maintenance time periods (MTP) – 5, 10, 15, 20 days – were considered in order to evaluate the algal growth dynamics.

Chlorophyll-a concentration

Model results for chlorophyll-a give frequent concentration values above 10 µg Chl-a/L, specially in river reaches downstream Barcelos, meaning high probability of eutrophication vulnerability. At Areias de Vilar WTP, lower concentration values are predicted (due to nitrification of organic matter discharged by Braga WWTP). This occurrence can have negative impacts on water quality of storage reservoirs to be built at the inlet of the planned WTP . Results for exceptional events with ten days time period, and for frequent dry-weather situations with twenty days maintenance time period are presented in Fig. 6.

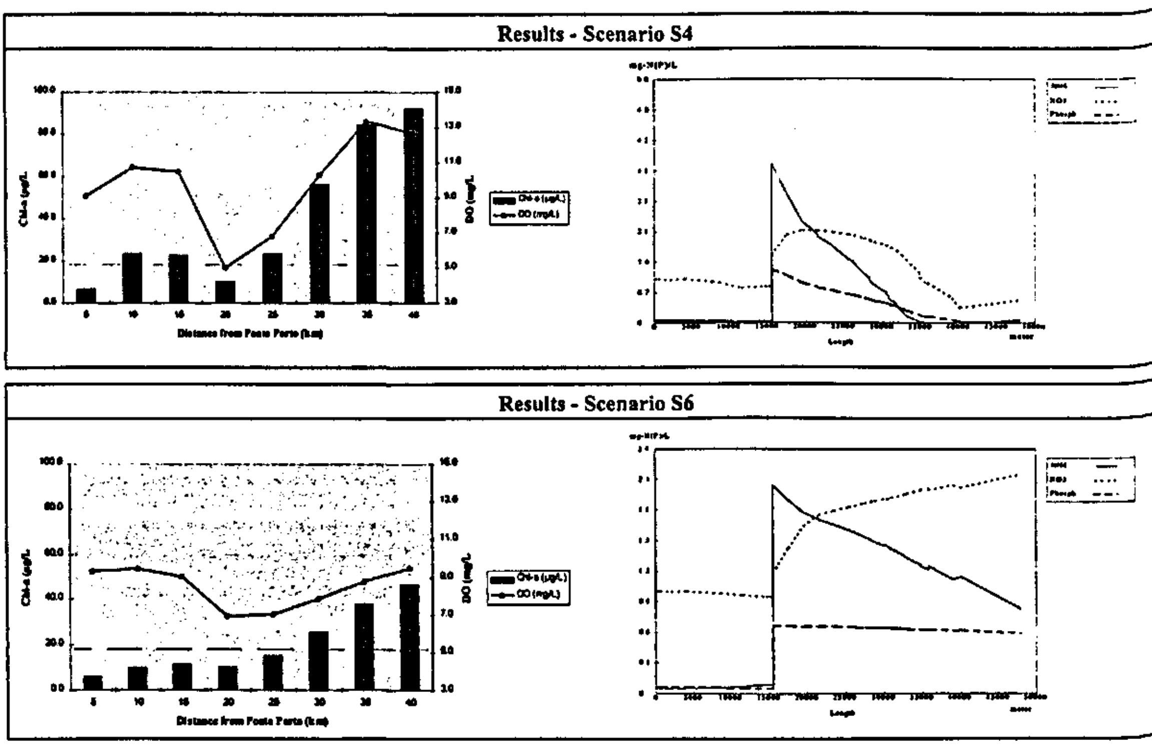


Figure 6. Chlorophyll-a and nutrients concentrations for exceptional and frequent dry-weather conditions.

Detailed results analysis at Areias de Vilar WTP has been done. Fig. 7 shows chlorophyll-a and DO concentration values for all of the scenarios simulated. Eutrophication critical situations can be anticipated in case of Braga WWTP failure associated with low flow river values (weekend operational regime and work break for maintenance purposes at Caniçada dam).

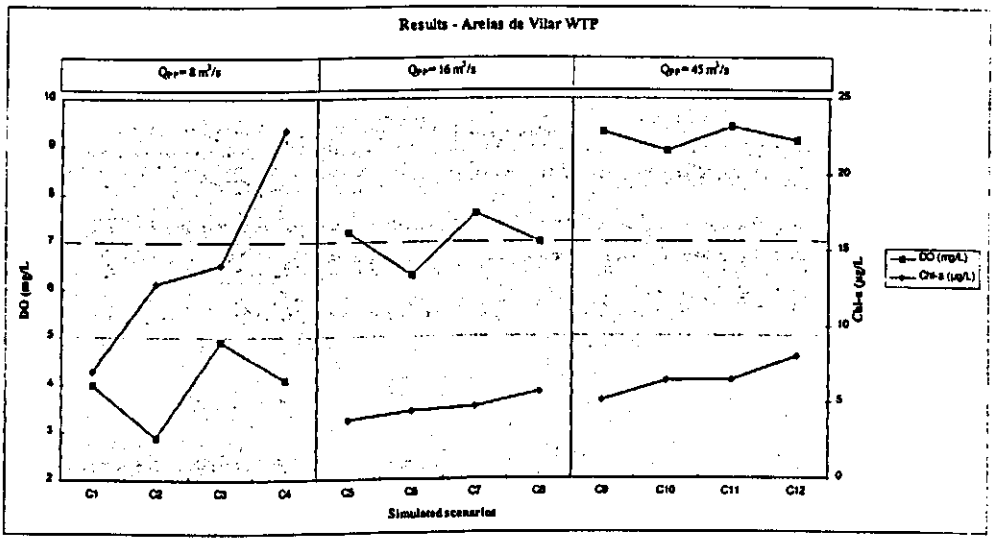


Figure 7. Global results for Areias de Vilar WTP location.

Algal Growth Dynamics

Short term analysis results for algae dynamics was worked out. Fig 8 shows chlorophyll-a spatial distribution for three MTP (5, 10, 20 days) under frequent dry-weather conditions (scenario S6).

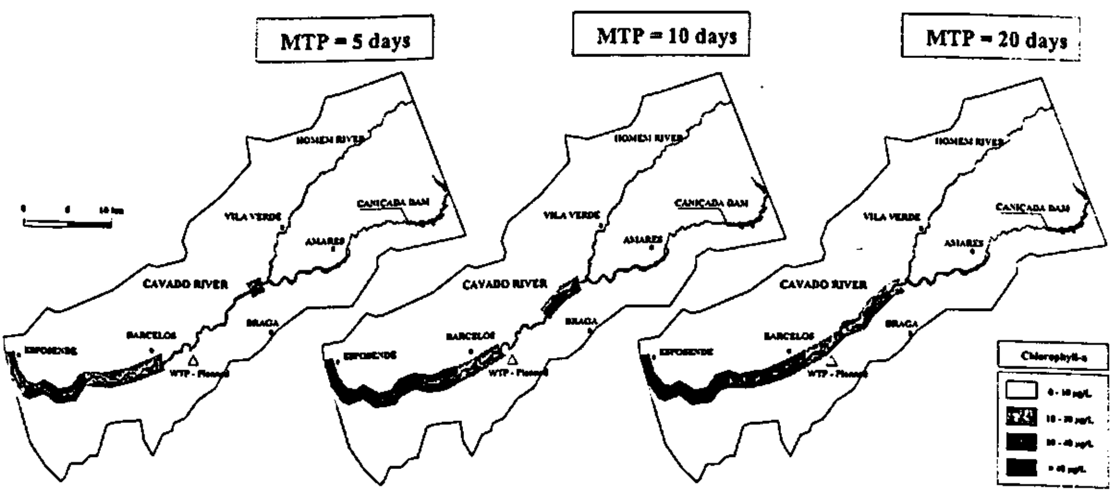


Figure 8. Algal growth dynamics for different time periods.

CONCLUSIONS

From eutrophication vulnerability analysis in river water, algae growth conditions can be anticipated at planned water abstraction point (Areias de Vilar). Actual concentrations strongly depends on water temperature, light energy and nutrients (N, P). Braga WWTP failure seams to be the major factor for river water eutrophication even when other nutrient sources are neglected.

This evidence suggests a cost-benefit analysis for alternative WTP abstraction point in a location upstream Braga WWTP effluent discharge. In fact, public health reasons (avoidance of pre-chlorination, and toxic compounds control difficulties) as well as technical-economical reasons (treatment process scheme simplification, and lower costs for alarm system, investment and operation costs) must be taken into account in order to define an integrated water management policy in the river basin.

REFERENCES

Abbott, M. B. (1979). *Computational Hydraulics Elements of the Theory of Free Surface Flows*, Pitman.

Betty N.G., Turner A., Tyler A.O. and Falconer R.A. (1996). Modelling Contaminant Geochemistry in Estuaries, *Water Research*, 30(1), 63-74.

Chau K.W. and Lee J.H. (1991). Mathematical modelling of Shing·Mun River network, *Water Research*, 14(3), 106-112.

ICIM (1992). *DUFLOW - A micro-computer package for the simulation of unsteady flow and water quality processes in one-dimensional channel systems*, Bureau ICIM, Rijswijk, The Netherlands.

PGIRHN (1992). *Impact assessment of water supply from the river Cavado. Preliminar studies*, Integrated Water Management Project for the North of Portugal, Porto (in Portuguese).

Thomann R. V. and Mueller J. A. (1987). *Principles of Surface Water Quality Modeling and Control*, Harper Collins.

University of Minho (1995). *River Cavado Water Quality Surveys*, Department of Biology (in Portuguese).

USEPA (1992). *EUTRO4 - Water Quality Analysis Simulation Program User's Manual*, Washington, D.C..

USEPA (1983). *Technical Guidance Manual for Performing Waste Load Allocations - Streams and Rivers (Nutrient/Eutrophication Impacts)*, Washington D.C.

Vieira J.M.P. and Lijklema L. (1989). Development and Application of a model for regional water quality management, *Water Research*, 23(6), 767-777.

Vieira J.M.P, Pinho J.L.S. and Duarte A.A.L.S. (1996). River Water Quality Management Using Mathematical Modelling, *Proc. Metropolitan Areas and Rivers*, 2, 258-270, ACEA Publisher, Italy.

MODELLING BENTHIC ACTIVITY IN SHALLOW EUTROPHIC RIVERS

W. Rauch* and P.A. Vanrolleghem**

*Department of Environmental Science and Engineering, Technical University of Denmark, Bldg. 115, 2800 Lyngby, Denmark
**BIOMATH Department, University Gent, Coupure links 653, 9000 Gent, Belgium

ABSTRACT

In this paper a simple modelling approach is presented that allows fast computation of benthic activity in rivers. The approach extends the half-order reaction concept used in biofilm models for use in a multiple substrate/multiple bacterial species system. Moreover, it is compatible with the IAWQ Activated Sludge Model n°1 format and has closed mass balances. The conversion of carbonaceous organic matter under aerobic and anoxic conditions and nitrification are represented in the model. The case study to which it was applied revealed that benthic activity is highly influenced by the eutrophic state of the river (presence or absence of algae). More specifically it was found that the spatial distribution of the species along the river is significantly different, resulting in postponed (downstream) nitrification in the eutrophied river. Also, oxygen depletion is found to be more severe and its spatial extension is larger.

KEYWORDS

Benthic activity, biofilm, eutrophication, half-order kinetics, modeling, river water quality.

INTRODUCTION

In river beds often a stable layer of organic material is found consisting of sediments, attached bacteria, algae etc. The conversion processes within this benthic biomass can have a considerable effect on the concentration of soluble compounds in the overlying water column, such as oxygen, organic matter and nutrients. Hence, benthic activity as the exchange of material between the water column and the benthos, is an important component of the processes responsible for water quality in the river, especially in shallow waters. Sediment oxygen demand, for example, may consume a significant amount of the available oxygen in the water phase (see Boyle and Scott, 1984; Rutherford et al. 1991; Horn and Wulkow, 1996). Furthermore, sediments can act both as a source and a sink for nutrients depending on the environmental conditions (aerobic, anoxic or anaerobic).

While the importance of benthic activity to the water quality is undoubted, the actual relations implemented in models to describe the phenomena are usually quite simple. Most often user specified fluxes such as the sediment oxygen demand are applied to predict the removal rate for the dissolved components in the bulk liquid above the benthos (see e.g. Bowie et al., 1985). Another simplified approach is to neglect the problem of mass transport inside the benthos. As a result, the conversions by the heterotrophic and autotrophic bacteria species inside the benthos can be calculated in the same way as being suspended, the only difference

being that the biomass is attached and the substrates (oxygen, organic matter and nutrients) are utilized from the water phase (see e.g. Reichert, 1994).
In the literature only very few models are found that apply the fundamentals of biofilm modeling to simulate the dynamics of the river benthos. This is mainly due to the complexity of the biofilm modeling approaches. Klapwijk and Snowgrass (1982) considered diffusion and zero order kinetics in three zones (aerobic, anoxic and anaerobic) in order to describe oxygen consumption, nitrification and denitrification in the sediments. The model has been applied mainly for large water bodies (case studies Hamilton Harbor and Lake Erie). Lau (1990) applied an analytical model to compute the effect of benthic activity to the removal of organic matter and oxygen consumption in open channel flow. The analytical solution of the second order differential equation describing diffusion and conversion inside the biofilm considers both zero - and first order reaction kinetics. Li and Chen (1996) also used an idealized description of the biofilm system but applied Monod - kinetics for the description of the bacterial metabolism. Hence, the equation had to be solved numerically by a trial and error method. Recently, Horn and Wulkow (1996) presented a model that describes the spatial variation of the conversion processes inside the biofilm in detail. The system of partial differential equations was solved numerically by means of a discrete Galerkin method. However, the model did not account for a dynamic development of the bacteria species in the biofilm and applied a constant fractionation of heterotrophic and autotrophic bacteria as well as a constant biofilm thickness.

In the following a simplified biofilm model is developed that aims at the description of the dynamics of the benthic biofilm and the influence of the conversion processes to the water quality in shallow eutrophic rivers. The model is based on an analytical solution that predicts the exchange of converted material between the bulk liquid and the biofilm, which is known as the half-order kinetic biofilm model (Harremoes, 1978). In this work, this approach was extended for description of the simultaneous or sequential conversion of multiple substrates such as readily biodegradable organic matter, ammonia, nitrate and dissolved oxygen by multiple bacterial species such as heterotrophic, nitrifying and denitrifying organisms. The dynamics of biofilm thickness are predicted on the basis of growth, decay, attachment and detachment and assumes constant biofilm density. One of the main features of this novel modeling concept is the fact that biochemical conversion processes can be described according to the background given in the IAWQ activated sludge model No. 1 (Henze et al., 1987). Moreover, being an analytical solution, a substantial gain in computational efficiency can be achieved.

MODEL DEVELOPMENT

Background

The main feature of biofilm modeling is that both mass transport due to diffusion and kinetic reactions have to be considered in one. The usual procedure is to derive a model that describes the system behaviour both in time and in space, the latter over the depth of the biofilm (e.g. Kissel et al., 1984; Wanner and Gujer, 1984). The basic idea of the simplified model presented here is to decouple the consideration of the diffusion process and the spatial distribution of bacteria species from the biokinetic reactions. This is done by means of a two - step procedure where (1) for each conversion process that is influenced by diffusion, the active fraction of the biomass within the biofilm is computed by means of a simple analytical solution to the problem and (2) all processes within the biofilm are then calculated as in a continuously stirred reactor but with only the active fraction of the species. This approach allows to describe biochemical conversion processes according to the background given in the IAWQ activated sludge model No. 1 (Henze et al., 1987) The concept requires the assumption of

- idealized spatial distribution of species
- ideal biofilm with homogeneous structure and density
- instantaneous steady state substrate profile
- absence of a stagnant liquid layer and

- absence of a temporal development of soluble components inside the biofilm.

Mass transport and 0-order reaction in a biofilm

Assuming an ideal situation, soluble substrate from the bulk liquid is transferred inside the biofilm and then transported further by means of molecular diffusion. The substrate is simultaneously utilized in the film by the bacterial species for growth. Harremoes (1978) developed the analytical solution for a steady state description of the problem:

$$\frac{\partial^2 S}{\partial x^2} = \frac{r}{D} \qquad (1)$$

where S = concentration of substrate at location x in the biofilm [ML^{-3}], D = diffusion coefficient for the substrate [L^2T^{-1}] and r = volumetric reaction rate in the biofilm [$ML^{-3}T^{-1}$]. To derive a solution to this second order differential equation the reaction rate needs to be defined. Harremoes (1978) pointed out that the specific growth rates of bacteria can be assumed zero order with respect to the concentration of the substrate S in the biofilm.

The reason is that the intrinsic saturation coefficients (assuming Monod type kinetics) are very small for the substrates at hand (dissolved oxygen, soluble organic matter, ammonia and nitrate). Hence, the biofilm volume where the 0 order kinetics assumption does not hold, is very small and can be conveniently neglected. On the basis of these considerations analytical solutions can be derived which are well known as half order kinetic models (Harremoes, 1978). Table 1 outlines default values for the diffusion coefficients.

TABLE 1: Default values for diffusion coefficients.

	Substrate	*cm^2/d*
D_{SO}	oxygen	2.1
D_{SNH}	ammonia	1.8
D_{SNO}	nitrate	1.6
D_{SS}	COD / BOD	0.6

General theory of active fractions

Applying a more refined concept (introduced by the IAWQ activated sludge model No 1 report; Henze et al., 1987) the reaction rate for the removal of soluble compounds from the water phase can be expressed also from the point of view of bacterial growth inside the biofilm. Equation 2 is a basic expression for a removal rate in a system with i substrates S and j species X. It specifies the volumetric reaction rate of a species with respect to each substrate:

$$r_{ij} = -\mu_j . X_j . \nu_{ij} \qquad (2)$$

where r_{ij} = zero order reaction rate for X_j with respect to S_i [$ML^{-3}T^{-1}$], μ_j = specific (max.) growth rate of species X_j [T^{-1}], X_j = bacterial species [ML^{-3}], ν_{ij} = stoichiometric coefficient [-], i = suffix denoting the substrates and j = suffix denoting the species.

The basic consideration which has to be made is whether the biofilm is fully penetrated by all substrates or not. In case the biofilm is fully penetrated (no substrate limitation) the solution to the problem is obvious, as all reactions take place over the full depth of the biofilm L with a constant (zero order) maximum rate. However, in case any substrate limitation occurs the reaction is taking place only over a certain depth of the biofilm and a partition occurs into (1) an active (upper) part and (2) an inactive part (close to the substratum). Since each reaction is governed by only one particular species, the limitation effect can be expressed by assuming only a certain fraction of this species to be active. Hence, a general solution can be given for the flux of each substrate into the biofilm:

$$J_i = \sum_j -\mu_{ij} . X_j . \nu_{ij} . \phi_j . L \quad \text{and} \quad \phi_j\,[0,1] \qquad (3)$$

where J_i = total transport of substrate i through surface of biofilm [$ML^{-2}T^{-1}$], L = biofilm thickness [L] and ϕ_j = active fraction of species X_j [-]

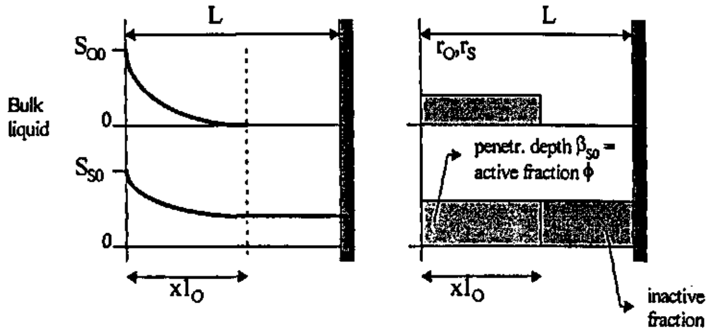


Figure 1: Illustration of a diffusion limited reaction. S_O is the limiting substrate.

Fig. 1 outlines the general idea of the theory to the simple example of a 2 substrate / 1 species system: For growth of heterotrophic bacteria in the biofilm two substrates are required, that is an electron acceptor (S_O dissolved oxygen) and an electron donor (S_S soluble organic matter). Both substrates are essential for bacterial growth and, consequently, the whole process stops when one of them is not fully available.

In this simple system the limitation of bacterial growth and the total conversion rate with respect to the substrate concentration in the liquid phase can be derived very easily. The underlying idea is that the dimensionless penetration depth of the limiting substrate β_{SO} (xl_O/L in Fig. 1) is also equal to the active fraction of the biomass ϕ. As the dimensionless penetration depth can be calculated analytically (Harremoes, 1978; Rauch, 1997) the fluxes of substrates into the biofilm are then derived directly from equation 3. However, note that the straightforward relation $\beta_{i,limiting} = \phi_j$ holds only for very simple systems as discussed in Fig. 2.

Indeed, a problem arises when this simple active fraction concept is applied to more complex problems due to the fact that, on the one hand, equations are given for the dimensionless penetration depth β_i that relate to the exhaustion of substrates, while, on the other hand, the active fraction ϕ_j for each biomass species is needed for further calculation. Although there exists a relationship between these two types of variables it is easily seen that this relation is case specific and requires a thorough analysis of the specific problem at hand, as illustrated below. A full description of the procedure is found in Rauch (1997). An application to wastewater treatment biofilm systems can be found in Rauch and Vanrolleghem (1997).

Concept for describing water phase - biofilm interaction

The procedure outlined above is actually only the first step for deriving a mathematical description of the biological conversion processes within the biofilm and of the mass exchange between bulk liquid and biofilm. The result of the analysis of the situation with respect to diffusion limitation is a quantification of the active mass of each bacterial species present in the whole biofilm . It is clear that also the dynamic changes in the bulk liquid need to be considered and the mass transfer between those two phases. The fact is that biofilm kinetics are directly connected with transport phenomena and, hence, it is not possible to put up a stringent separation of biokinetic and physical processes.

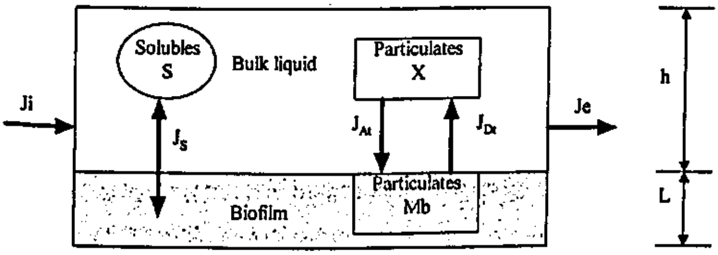


Figure 2: Interaction processes between biofilm and bulk liquid components due to mass fluxes J_i.

The whole system is seen in the following as two connected continuously stirred tank reactors where one is representing the water phase and the other the biofilm (Fig. 2). The components in both tanks are expressed differently, in the water phase in terms of concentrations $[ML^{-3}]$,as usual, and in the biofilm as mass M_b [M].

The reason is that the thickness of the biofilm compartment is constantly changing, which can be taken into account more easily by balancing masses than by concentrations. Neglecting the dynamic changes in biofilm density the thickness of the biofilm is computed at each time instant from

$$L = \frac{\sum M_{bi}}{\rho^{*}.A_0} \tag{4}$$

where ρ^* = constant mass of dry biomass per wet biofilm volume $[ML^{-3}]$, A_0 = surface of biofilm $[L^2]$ and M_{bi} = mass of particulate component i in the biofilm [M]

The different dimensions of the particulate components in both reactors (concentrations in the bulk liquid and mass in the biofilm) do not allow a direct conversion of components in both compartments. The formulation of the mass transfer between the phases has to account for that. However the volumetric reaction rate r_{Vi} with respect to the substrate concentration in the bulk liquid can also be expressed as

$$rVi = \sum_j -\mu j.vij.\phi j.\frac{Mbj}{V} = \sum_j -\mu j.vij.\phi j.Xbj \tag{5}$$

where V = volume of bulk liquid compartment $[L^3]$, $X_{bj} = M_{bj}/V$ concentration of particulate matter j in the biofilm per unit of volume of the bulk liquid compartment $[ML^{-3}]$. Hence, the problem of the different dimensions of the particulate components in both phases is taken care of directly and must no longer be considered for description of the mass transfer between bulk liquid and biofilm.

Physical interaction between biofilm and bulk liquid

Attachment (flux J_{AT}) is addressing a number of physical processes where suspended matter is transported from the water phase to the biofilm compartment. The most important phenomenon is sedimentation which is generally described as a first order process with respect to the concentration of the particulate matter in the water phase. The reverse process of displacement is addressed as detachment (flux J_{dt}). Detachment describes the material loss from the biofilm matrix and is usually categorized into the phenomena erosion, sloughing and abrasion. In the following no distinction is made between these three phenomena as it is felt that the detailed processes significantly lack understanding. Detachment is assumed to be proportional to the friction forces onto the surface of the biofilm as well as to the material mass. Hence, the process is described as being proportional to the product of mass of particulate matter in the biofilm and flow in the water phase.

Description of biokinetic processes in the biofilm - Process matrix

The biokinetic process description is straightforward once the fractions of the active biomass have been computed as previously outlined. Expressing the components in the biofilm as above in terms of concentrations with respect to the volume of the bulk liquid compartment (X_{bj}) does not violate mass conservation principles. In the following aerobic and anoxic growth of heterotrophs and autotrophs is simulated, as well as hydrolysis and decay. Although the description of the biokinetic processes in the biofilm follows as closely as possible the concept of ASM1 some simplifications had to be implemented. First of all bacterial growth is not expressed as a Monod type reaction as done in ASM1 but instead as a first order process with respect to the active fraction of the bacterial mass alone. This is due to the requirements for zero order in substrate kinetics and diffusion limitation. In ASM1 also the limitation of reactions with respect to oxygen and ammonia is expressed by Monod type switching functions. Conveniently these functions can be dropped as all limitations are already considered in the active fractions.

It is postulated that hydrolysis is a first order process with respect to the substrate concentration. Furthermore, it is assumed that readily biodegradable organic matter from hydrolysis is instantaneously transferred into the bulk liquid. This assumption might be a rather crude simplification of reality, however, the fact is that we still do not know enough about this process in order to make a better funded statement.

TABLE 2: Process matrix for biokinetic processes in the biofilm and corresponding effect to the concentration of soluble components in the bulk liquid. Below: default values used in the case study.

Process	S_O ML^{-3}	S_S ML^{-3}	S_{NO} ML^{-3}	S_{NH} ML^{-3}	X_{bH} ML^{-3}	X_{bA} ML^{-3}	X_{bS} ML^{-3}	X_{bI} ML^{-3}	Process rate $ML^{-3}T^{-1}$
aerobic het. growth	$1-1/Y_H$	$-1/Y_H$		$-i_x$	1				$m_H.X_{bH}.f_H$
anoxic het. growth		$-1/Y_H$	$-(1-Y_H)/2.86Y_H$	$-i_x$	1				$m_{HH}.X_{bH}.f_H^*$
aerobic aut. growth	$1-4.57/Y_A$		$1/Y_A$	$-1/Y_A-i_x$		1			$m_A.X_{bA}.f_A$
decay het.					-1		1-fp	fp	$b_H.X_{bH}$
decay aut.						-1	1-fp	fp	$b_A.X_{bA}$
hydrol.		1		i_x			-1		$kh.X_{bS}$

Stoichiometric Parameters			Kinetic Parameters		
Y_H	=	0.20	m_H	=	3.00 (d^{-1})
Y_A	=	0.06	m_{HH}	=	1.50 (d^{-1})
ix	=	0.08 (gN/gCOD)	m_A	=	0.15 (d^{-1})
fp	=	0.08	b_H	=	0.20 (d^{-1})
			b_A	=	0.03 (d^{-1})
			kh	=	1.00 (d^{-1})

CASE STUDY

The applicability and realism of the model was tested in a virtual case study where the temporal and spatial impacts of a constant discharge of municipal wastewater is evaluated in a shallow eutrophic river. The system is a steady state system (Table 3 outlines the physical properties) with only the eutrophication being dynamic and following a diurnal pattern. It is assumed that (in this case) the benthic activity in the river has an overwhelming effect to the degradation of soluble compounds in the water phase and, hence, the influence of suspended bacteria is negligible. Consequently, also the bacterial biomass discharged in the treatment plant effluent has been considered here as being slowly biodegradable organic matter, which is subject to degradation, and not as a pelagic bacteria compound (see also Rauch and Harremoes, 1996).

TABLE 3: Constant system properties	Natural River	WWTP Effluent
Flow (m^3/s)	0.04	0.01
Velocity (m/s)	0.25	
Width (m)	2.0	
S_S ($gCOD/m^3$)	0.0	1.0
X_S ($gCOD/m^3$)	0.0	20.0
S_{NH} (gN/m^3)	0.0	10.0
S_{NO} (gN/m^3)	0.0	10.0
Temperature (°)	15	15

Algae and rooted macrophytes cause a significant diurnal variation in the oxygen concentration due to photosynthesis and respiration. This eutrophication effect was here taken into account by means of dynamic forcing functions as described in Harremoes (1990). Expressed in aerial units the maximum photosynthesis is 10.0 $gO_2m^{-2}d^{-1}$, the respiration rate 5.0 $gO_2m^{-2}d^{-1}$ and the reaeration coefficient k_2 is 10.0 d^{-1}. Reaeration is modeled according to the literature as a first order process (Bowie et al., 1985). The longitudinal transport processes of soluble compounds in the water phase is described by a box model, i.e. by interlinked mixed reactors (Rauch et al., 1997).

In Fig. 3 predicted oxygen and ammonia concentrations are plotted for a dynamic steady state of the system, meaning that the system shows a recurring photosynthesis induced diurnal variation. Profiles are given at different times of the day for the eutrophic system described above and for a reference non-eutrophied situation, where photosynthesis/respiration are absent. The interesting aspect is that the DO in the water phase downstream of the discharge significantly differs. Although photosynthesis and respiration only cause a sinusoidal variation of oxygen (ΔDO= 4.0 gO/m^3) around the mean value of DO_{mean} = 10.0 gO/m^3 the effect of a presence of photosynthesis/respiration on the oxygen consumption by the benthic activity is stunning: The minimum DO concentration in the river is significantly lower and also the spatial extension of the depletion is larger as compared to the reference situation. A similar difference can be observed in the higher ammonia concentrations observed over longer distances along the river stretch (Fig. 3 right).

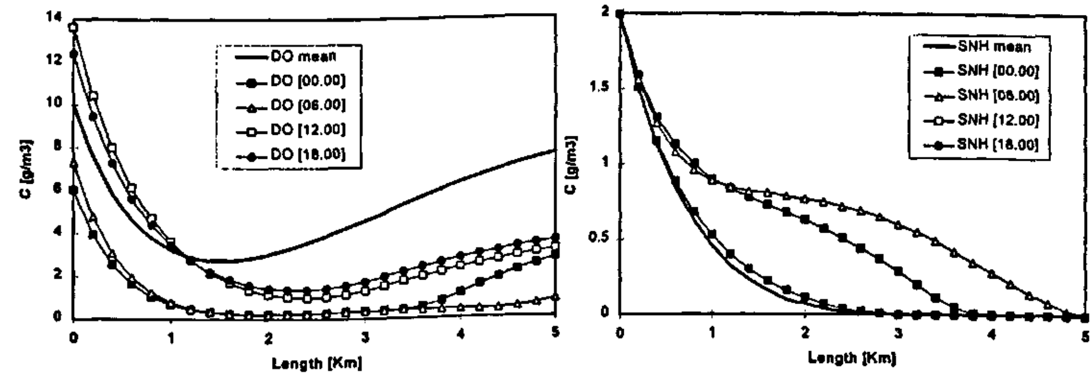


Figure 3: Steady state oxygen (DO) and ammonia (SNH) profiles in the eutrophic river at different times of the day and for a reference situation, i.e. for a constant mean oxygen concentration (DO mean) in the river upstream of the discharge point.

To explain this difference, Figure 4 (right) outlines the situation: Both the heterotrophic and the autotrophic bacterial species compete for substrate (DO) and space inside the biofilm. The slowly biodegradable organic matter (X_S) in the municipal wastewater discharge has to settle first and is only then converted to easily biodegradable organic matter (S_S) by hydrolysis. Consequently, immediately downstream of the discharge point more substrate is found for nitrifiers (S_{NH}) than for heterotrophs (S_S). Hence, there is a larger fraction of nitrifying bacteria in this part of the river. More substrate is available for the growth of heterotrophic biomass further downstream of the discharge point. Accordingly, the fraction of nitrifiers in the benthic biomass rapidly declines in this river stretch to the favor of the heterotrophic bacteria. Note that the fraction of inert solids in the biofilm (1-f(MH)-f(MA)) increases downstream as the substrate loading decreases.

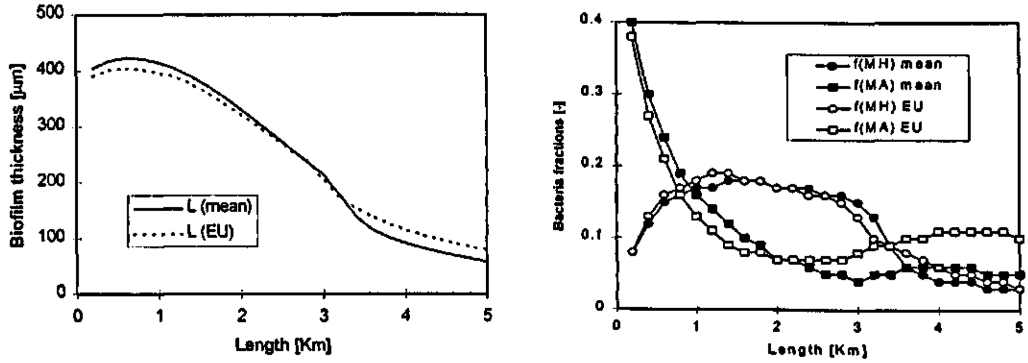


Figure 4: Steady state profiles for the situation with DO fluctuations (EU) and without (mean).
Left: Biofilm thickness in µm and Right: fractions of heterotrophic (MH) and autotrophic (MA) biomass in the benthos.

Moreover, this competition of the bacterial species is also the reason for the large differences found in the systems' behaviour between the eutrophic and non-eutrophic situation: As the heterotrophic bacteria can adapt much easier to changing environmental conditions they have an advantage in the competition for substrate (DO) and space under natural DO fluctuations. Hence, the fraction of nitrifiers is slightly reduced in the first 2 km after the municipal wastewater discharge as compared to the non-eutrophic situation.

Between km 2 and 5 downstream of the discharge the amount of substrate available for the heterotrophs gets sparse (both S_S and X_S are removed from the bulk liquid) and nitrifying bacteria grow more abundant. Here the situation with respect to the nitrifying bacteria is reverse as the fraction is higher in case of eutrophication as in the reference situation. The reason is clear from Fig 3 right: In case of DO fluctuations less ammonia is degraded in the first 2 km and, therefore, substrate for growth is still available. This behaviour is also reflected in the differences in the biofilm thickness profile (Fig 4 left) calculated with a biofilm density ρ^* being 60 kg COD/m^3. Concluding, in the presence of photosynthesis and respiration activity nitrification occurs further downstream from the point of discharge as compared to the non-eutrophic reference situation.

CONCLUSIONS

In this paper a conceptually simple numerical model for the description of the dynamics of the benthic activity is developed. The model builds on the concepts of biofilm modeling and relies also on the background given in the IAWQ activated sludge model No. 1. This approach ensures compatibility with the state of the art in describing biochemical conversion processes. Substrate utilization of carbonaceous matter, nitrification and denitrification is taken into account as well as hydrolysis of attached organic material. The main advantage of this new approach is seen in the reduced complexity of the model compared to classical biofilm modeling, which allows fast computation and, therefore, the application of the model for simulating the impact of benthic activity on water quality in rivers. The case study reveals that the development of bacterial species is very sensitive to eutrophication. A significant fluctuation of the dissolved oxygen concentration in shallow rivers due to photosynthesis and respiration is likely to yield a competition of the different bacteria species in the benthos. In this case, the result was a severe detrimental impact to the oxygen profile downstream of the discharge of municipal wastewater as compared to a non-eutrophic situation.

ACKNOWLEDGEMENT

This work was financially supported by the Fund for Scientific Research (F.W.O.), Belgium and by the Austrian government.

REFERENCES

Bowie G., Mills W., Porcella D., Campbell C., Pagenkopf J., Rupp G., Johnson K., Chan P., Gherini S. and Chamberlain C. (1985): Rates, constants and kinetics formulations in surface water quality modeling. 2nd Ed. US EPA/600/3-85/040.

Boyle J. and Scott J. (1984): The role of benthic films in the oxygen balance in an East Devon river. Wat. Res., 18, 1089-1099.

Rutherford J.C., Wilcock R.L and Hickey C.W. (1991): Deoxygenation in a mobile-river bed. Wat. Res., 25, 1487-1497.

Harremoes P. (1978): Biofilm kinetics. in Mitchell (ed.): Water pollution microbiology, 2, 71-109, Wiley N.Y.

Harremoes P. (1990): Water pollution. Polyteknisk Forlag. Lyngby, Denmark.

Henze M., Grady C.P.L., Gujer W., Marais G.v.R. and Matsuo T. (1987): Activated sludge model No. 1. IAWQ, London.

Horn H. and Wulkow (1996): Modellierung von Einleitungen in kleine Fliessgewaesser. gwf Wasser-Abwasser, 137, 10, 557-564.

Klapwijk A. and Snodgrass W.J. (1982): Biofilm model for nitrification, denitrification and sediment oxygen demand in Hamilton Harbor. Paper presented at 55th annual Water pollution Control Federation Conference, St. Louis, MO, Oct 2-6.

Kissel J.C., Mc Carty P.L. and Street R.L. (1984): Numerical simulation of a mixed-culture biofilm. J. Env. Engin., 110, 393-412.

Lau Y.L. (1990): Modeling the consumption of dissolved contaminants by biofilm periphyton in open-channel flow. Wat. Res., 24, 1269-1274.

Li S. and Chen G.H. (1994): Modeling the organic removal and oxygen consumption by biofilms in an open-channel flow. Wat. Sci. Tech., 30(2), 53-61.

Rauch W. (1997): A dynamic biofilm model for simulation of wastewater treatment processes and benthic activity in rivers. Technical report. BIOMATH Department, University Gent, Belgium.

Rauch W. and Harremoes P. (1996): The importance of the treatment plant performance during rain to acute water pollution. Wat. Sci. Tech., 34(3-4), 1-8.

Rauch W. and Vanrolleghem P.A. (1997): A simple dynamic multispecies biofilm model which considers diffusion and degradation of multiple substrates. In: Proceedings 11th Forum Applied Biotechnology. Med. Fac. Landbouww. Univ. Gent, 62 (in Press).

Rauch W., Henze M., Koncsos L., Reichert P., Shanahan P., Somlyódy L. and Vanrolleghem P. (submitted). River Water Quality Modeling: State of the Art. Submitted to *Wat. Sci. Tech.*

Reichert P (1994): Mathematische Modellierung aquatischer Systeme. Lecture manuscript EAWAG, Duebendorf, Switzerland.

Wanner O. and Gujer W. (1984): Competition in biofilms. Wat. Sci Tech., 17, 27-44.

TREND ANALYSIS OF EUTROPHICATION IN DUTCH COASTAL WATERS FOR 1976 THROUGH 1994 USING A MATHEMATICAL MODEL

F.J. Los[1], M. Bokhorst[1], I. de Vries[2] and R.J. Jansen[2]

[1]) Delft Hydraulics / WL, Marine and Coastal Management, p.o. box 177, Rotterdamseweg 185, 2600 MH Delft, The Netherlands
[2]) National Institute for Coastal and Marine Management / RIKZ, p.o. box 20907, 2500 EX The Hague, The Netherlands

ABSTRACT

In the last decades a number of measures have been taken to reduce the nutrient discharges to the western European surface waters. As a result the phosphorus concentrations of several major European rivers, the Rhine in particular, have decreased. Locally this has resulted in a noticeable reduction of algal biomasses in general and of blue green algae in particular. The anthropogenic phosphorus discharges into the Dutch part of the North Sea have also declined recently. However the impacts on the North Sea ecosystem have so far been limited. Time series analysis reveals a significant decreasing trend for dissolved inorganic phosphorus (40%) whereas chlorophyll, an indicator for total algal biomass, shows a small and mostly not statistically significant decreasing trend (de Vries, 1997). To explain these observations the National Institute for Coastal and Marine Management and the North Sea Directorate of the Dutch Ministry of Public Works have committed Delft Hydraulics to reproduce the observed trends, the spatial gradients and long term seasonal patterns for the Dutch coastal zone for a 20 year period from 1975 through 1994 with a detailed coupled physical-ecological model. The model results indicate a small effect of decreasing phosphorus but an important role of both nitrogen and light climate for primary production.

OVERVIEW OF THE MODELLING INSTRUMENT

About ten years ago the first Dutch eutrophication models were developed for various parts of the North Sea. In this first generation of models processes were described by rather simple equations. Transports were based on 2D hydrodynamic calculations. As a compromise between computational performance and accuracy a uniform grid size of 16x16 km was selected (the so called GENO grid). For this application to the southern Bight of the North Sea this model includes 1395 computational elements. With the water quality model, named DYNAMO, it was possible to obtain reasonable results for total algal biomass (expressed in chlorophyll conentrations) and nutrients. With this model it was not possible to compute the composition of the phytoplankton bloom, variations in internal characteristics (stochiometry) such as nutrient to biomass ratios and steep gradients due to temporal or spatial variations.

Algal blooms usually consist of various species of phytoplankton belonging to different taxonomic or functional groups such as diatoms, microflagellates and dinoflagellates. They have different requirements for resources (nutrients; light) and they have different ecological properties. Some species are considered to be objectionable for various reasons. Among these are *Phaeocystis*, which causes foam on the beaches and various species of dinoflagellates, which among others may cause diuretic shell fish poisoning. To deal with these phenomena it is necessary to distinguish different types of phytoplankton in a model.

For Dutch fresh water systems the development of eutrophication models had started about ten years earlier, in the late 1970s. Much of this work was performed at Delft Hydraulics for both the national and for several local governmental authorities. These experiences were formalised in the DELWAQ-BLOOM-SWITCH (DBS) model (Delft Hydraulics, 1992). This model and some of its predecessors have been successfully applied to many different systems with great differences in physical and chemical conditions. They range in depth from 1 to 30 m, in extinction from 0.5 to 10 m-1, in residence times from about 1 day to several years. Nutrients conditions vary from meso- to hyper eutrophic, chlorophyll peaks range from 1 to 500 μg/l, occasionally even more. Compared with DYNAMO, many processes are described in greater detail in this model. The performance of this fresh water modelling instrument was thoroughly validated against numerous sets of observations (Delft Hydraulics, 1991; Delft Hydraulics, 1994)

Phytoplankton within DBS is computed by the module BLOOM II. It is based upon the principle of competition between different species. The basic variables of this module are called types. A type represents the physiological state of a species under strong conditions of limitation. Usually a distinction is made between three different types: an N-type for nitrogen limitation, a P-type for phosphorus limitation and an E-type for light energy limitation. The solution algorithm of the model considers all potentially limiting factors and first selects the one which is most likely to become limiting. It then selects the best adapted type for the prevailing conditions. The suitability of a type (its fitness) is determined by the ratio of its requirement and its growth rate. This means that a type can become dominant either because it needs a comparatively small amount of a limiting resource (it is efficient) or because it grows rapidly (it is opportunistic). The algorithm considers then the next potentially limiting factor and again selects the best adapted phytoplankton type. This procedure is repeated until it is impossible to select a new pair of a type and limiting factor without violating (i.e. over-exhausting) some limiting factor. Thus the model seeks the optimum solution consisting of n types and n limiting factors. As a further refinement BLOOM takes the existing biomasses of all phytoplankton types into account. These are the result of production, loss processes and transport during the previous time-steps. So the optimisation algorithm does not start from scratch. As they represent different stages of the same species, the transition of one type to another is a rapid process with a characteristic time-step in the order of a day. Transitions between different species is a much slower process as it depends on mortality and net growth rates. It is interesting that the principle just described, by which each phytoplankton type maximises its own benefit, effectively means that the total net production of the phytoplankton community is maximised. Several modules from this fresh water model system were gradually included in the salt water model to obtain a greater accuracy and new output variables such as a further distinction of the algal community into major functional groups or even species. The marine version of BLOOM, called North Sea BLOOM, considers diatoms, flagellates, dinoflagellates and *Phaeocystis*.

CALIBRATION PROCEDURE

Methodology Selecting a methodology for the calibration of a complex model for a complex system is not trivial. First of all the results for about half a dozen of different substances have to be considered. Second seasonal and regional variations must be taken into account. For the trend analysis project chlorophyll (as an indicator for total biomass), dissolved nutrients (nitrate, ortho-phosphorus, silicate), the distinction between different groups of phytoplankton (only qualitatively) and the extinction of light (little data) have been evaluated. To check the hydrodynamics also observations for the salinity have been used. Model outputs were compared to observations for about 20 different stations arranged in several transects. The locations are shown in Figure 1. For some of these stations yearly seasonal data are available for 1974 through 1993. For most stations some years are lacking, but in general data availability is unusually large. For the calibration of previous versions of North Sea BLOOM data for 1985 were used. In recent years some changes were observed due to the aforementioned reduction in phosphorus discharges. Moreover in the international political arena, 1990 is a milestone. Thus 1990 was selected for the calibration of the application of North Sea BLOOM.

Hydrodynamics Size, arrangement and transport between computational elements was derived from a 3-D hydrodynamic model of the Dutch coast. From these a 2D vertically averaged schematisation was determined for the water quality model. The water quality model consists of over 2000 elements with varying sizes of about 1x1 km near the coast up to a few square kilometres at the model boundaries. With the underlying hydrodynamic model eleven representative situations were computed for different combinations of wind speed and direction. An attempt was made to simulate actual transports by interpolation between the eleven flow fields. However, computations with BLOOM showed that the results lacked realism. For instance at the Noordwijk 10 station spring values of nitrate were computed for the calibration year 1990 that were clearly outside the range that was observed over the entire 20 year period. Instead a different approach was adopted. The residual flows computed for the average south western winds were used, but with a dispersive correction factor for other wind directions. So if in a particular period during a particular year the wind is for instance north west, the horizontal dispersion perpendicular to the coast is increased. This approach proved to be quite successful in reproducing observations.

Nutrient loading Data for nutrient loading for the entire 20 year period were collected by the National Institute of Coastal and Marine Management. The most important source of nutrients is the river Rhine. For the calibration of the model the 1990 measurements were used to estimate the nutrient loading.

Sea boundaries The model area has open boundaries to the south, west and north (See Figure 1). Since the prevailing transport direction along the Dutch coast is from south west to north east, the southern boundary is by far the most important. Concentration values here were derived from the measurements at the nearby Appelzak transect. Values for elements for which there are no measurements, were obtained by fitting an exponential curve through the observations. Little variations are observed along the western boundary. Therefore concentration values here were obtained by simply taking the average of all observations at all stations 70 kilometres off the coast. Concentrations at the northern boundary were obtained from measurements at the Rottum transact. Notice that this is by far the least important boundary due to the prevailing direction of the currents. Most of the model substances are measured directly and the observed data were directly used as boundary concentrations. For other substances such as the biomasses of different types of phytoplankton some conversion functions were needed as there are no direct measurements. These functions are based on the experience obtained in numerous studies for both fresh and salt water systems.

Model forcing Data for the solar radiation, wind speed and wind direction were obtained from the Royal Dutch Meteorological Institute. Data for the water temperature were provided by the National Institute of Coastal and Marine Management. As there are little variations within the model area, the same forcing is used for all computational elements.

Light attenuation Light is an important limiting factor for phytoplankton in Dutch coastal waters. Determining adequate values for the light attenuation factor is therefore important for the model performance. Originally the total contribution of inorganic material was computed as the sum of the background extinction coefficient and the extinction due to suspended matter, which is computed by a sub-model. Seasonal variations are (simply) computed using a cosine function. This approach proved to be adequate in reproducing the observed suspended matter data. It was, however, impossible to compute correct values for the extinction contribution due to non algal material using these suspended values and a single, uniform conversion coefficient. In particular in regions with large fresh water discharges the extinction coefficient was under estimated. Hence the model would compute algal biomasses exceeding the observations. Using a larger specific coefficient for suspended matter would improve the model's results near the coast, however at the expense of off-shore regions where the inorganic fraction of the extinction coefficient would be too large. Thus an additional term was introduced into the equation to compute the extinction based upon the local salinity values. This was previously proposed for the Eastern Scheldt (Peeters et al., 1991). The underlying assumption is that fresh water contains various substances such as humic acids contributing to the extinction. The coefficients by which the salinity and suspended matter are multiplied have been determined by calibration using observations on the extinction coefficient from the EUZOUT project for 1987 through 1990 (Peeters et al., 1991).

Water quality and ecological processes At the beginning of the project a calibrated North Sea BLOOM version existed for the GENO grid. The set of processes and their corresponding coefficient values were used as the starting point for the model calibration. In general the appropriate value for a model coefficient must always be regarded in relation to the process formulations adopted by that model. In other words the coefficient values will always compensate some of the inaccuracies of a model. So when the moderately refined GENO grid was replaced by the much more detailed grid used in the trend analysis study it was conceivable that many changes to the coefficients for the water quality processes would be necessary. Fortunately preliminary results showed that only some minor changes were necessary.

Light dependency So far it was assumed that the light dependency of growth was best described by an optimum curve as is often observed in the laboratory. Hence growth declined at intensities exceeding the optimum value (photoinhibition). From a further analysis of the mixing patterns and light intensities it became obvious that inhibition will be a rare phenomenon in the Dutch coastal waters considered here: in general phytoplankters simply do not spend a sufficient amount of time at high intensities. Therefore the light dependency curves have been adapted, maintaining the initial part, but now assuming the growth efficiency to remain equal to 1.0 at the optimum and all higher intensities. Due to this modification the average growth rates increase, which is particularly important at locations where light limitations frequently occur. With the old curves considering photo-inhibition, the maximum biomasses in relatively shallow locations near the coast could not be reproduced as they proved to be considerably below the observations.

Maximum growth rates In relation to the changes in the light response curves some minor adjustments were also made to the maximum growth rates of the species groups considered in North Sea BLOOM. In general, maximum growth rates were somewhat decreased. This is true in particular for dinoflagellates. Literature sources indicate that a further reduction might still be necessary, but within the current model set-up this would practically eliminate this group from the output.

Stochiometry The combination of the growth rates and the stochiometric coefficients determine which type will dominate under each set of conditions. To improve the selection of types, some minor adjustments with respect to the previously used coefficient values have been made. These modifications have no significant effect on the total biomass, only on its composition.

Mortality and sedimentation Previously diatoms were the only algea group which were considered to settle. Observations indicate that in general, algal biomasses often decline rapidly immediately following a bloom. There is some debate on the mechanisms, however. It is often assumed that the grazing pressure increases (external cause). As an alternative assumption the vitality of phytoplankters might decline under nutrient stress due to a degradation of their fitness (internal cause). Perhaps both mechanisms are important. What ever the mechanism, the result is a rapid decline in biomass and an enhanced production of detritus. This effect can be easily included in the model by a change in the net sedimentation rate of nutrient limited types. During the calibration procedure a settling rate 0.5 m/day has been established. This value is used for all groups except dinoflagellates, as sedimentation is probably insignificant for this group with active buoyancy regulation. Compared to previous model versions concentration peaks become steeper, first of all due to the enhanced sedimentation rate when nutrients get depleted but secondly because a larger fraction of the nutrients is now mineralised at the bottom rather than in the water column. Since degradation processes proceed at a lower rate in the bottom compared to the water phase, there is shift towards the end of the season in the recycling of nutrients. As a result, biomasses during summer tend to be lower in nutrient limited locations, but distinct autumn peaks are now a more common phenomenon in the model. In general this behaviour compares better with measurements.

CALIBRATION RESULTS

There are numerous ways to consider the results of a complicated model such as North Sea BLOOM because variations in both space and time for a large number of substances and fluxes might be considered. Here three types of outputs are presented:

1. Times series of some important substances at two representative locations,
2. Gradients perpendicular to the coast for two substances at one transect,
3. Geographical plots for some important substances at three representative times.

Time series A large number of potential locations and years were available for the calibration of the model. Only the model results for 1990 have been used during the calibration and a careful comparison between model outputs and observations has been made for all locations monitored during this particular year. The substances that we considered have been mentioned in "Methodology". The 1990 model results are presented against the mean and median of the observations over the period 1975-1993. To give an indication of the temporal variations the 16.6% and 83.3% quantils are shown.

Figure 3 shows the time series for the location Noordwijk 10, where nutrient levels are so high that light energy is the most important limiting factor. Both the calculated concentration of about 20 µg/l and calculated timing of the chlorophyll spring peak agrees well with the measurements. Computed summer values tend to be high in comparison to the observations. Computed levels of nitrate start at about 0.60 mg/l and achieve a maximum value in March of about 1.0 mg/l, after which they decline rapidly due to uptake by phytoplankton. They remain low during the growing season until August and rise to about 0.60 mg/l by the end of the year. The agreement with the observations is excellent. Computed and observed ortho phosphate levels agree well, but in contrast to nitrate (and silicate) the model tends to under estimate the summer values. The main reason for this discrepancy is most likely the lack of inorganic phosphate adsorption and desorption in the present model version (See "Conclusions and future developments"). Computed and observed levels for silicate are very similar to those for nitrate. The main difference is the period at which silicate is virtually depleted due to uptake by diatoms: a period of about 100 days (from May to August).

Figure 4 shows the results for the location Terschelling 4, where all three macro nutrients get depleted during the growing season. The computed chlorophyll peak is of the same magnitude (20 µg/l) and occurs earlier compared to Noordwijk 10. Notice that a clear autumn peak is computed in August In general the computed levels agree well with the observations. Computed and observed nitrate levels are about 40 percent below those at Noordwijk 10. As a result this nutrient is now limiting for a period of almost 100 days. Computations reflect measurements accurately. There is considerable variation in the measured phosphate levels at his station. In winter computed and measuremed levels agree well. The observed summer levels are under estimated by the model. There is an excellent agreement between computed and measured levels of silicate at this station.

Gradients perpendicular to the coast With the Dutch coastal waters the concentrations of various substances may vary an order of magnitude over a distance of about 50 km. This is due to several factors: (1) large nutrient discharges from the rivers Rhine and Meuse which remain in a 'plume' along the coast, (2) a large increase in depth from about 5 m adjacent to the coast up to about 20 m at 10 km off shore and (3) high suspended matter concentrations and hence a high turbidity at locations near the coast. The regions at which these phenomena manifest themselves are not identical, but do show considerable overlap. The resulting gradients lead to variations in concentrations over distances of only a few km. To simulate these by a model, the grid size must be of the same order of magnitude, which is the case in the coastal waters model. As an example of the model performance the results for chlorophyll and ortho phosphorus at the Egmond transect in May 1990 are presented (Figure 2). Together with the model results the average, maximum and minimum value of the observations from 1975 up to 1993 are shown. It may be concluded that the model results are close to the averages of the observations for both substances. This is even true for the first few km offshore, where the level of variation in the measurements is relatively large.

Geographical images The model results for individual locations or for a transect at a single time-step compare well with the observations and show that a broad range of conditions can be adequately reproduced by the model. To illustrate the spatial behaviour of the model a number of characteristic images for chlorophyll, the main limiting factors and the species groups will be presented. The time steps selected for presentation are begin April during the spring bloom, begin June in the midst of summer and begin August during the autumn bloom.

Spring peak (Figure 5) Chlorophyll levels are relatively high during the spring bloom varying from 4 up to 50 µg/l in some regions. Light energy is by far the most important limiting factor; there is no nitrogen limitation, and phosphorus limitation is confined to a small region. A complex interplay of depth, transport and suspended matter leads to a biomass pattern in Dutch coastal waters where biomass levels are highest adjacent to the coast and in some regions between 20 and 40 km off-shore. Concentrations are comparatively low at locations 10 km offshore because here the background extinctions are generally high (suspended matter and fresh water) and the depths are already in the order of 20 meters. The model biomass mainly consists of diatoms and *Phaeocystis*; flagellates and dinoflagellates are virtually absent.

Summer situation (Figure 6) Two months later during the summer the situation has changed drastically. Light limitation is now confined to areas with high suspended matter concentrations. These are the erosion areas in front of the coast of Zeeland (relatively low nutrient concentrations) and the plume of the river Rhine (high nutrient concentrations). Everywhere else nitrogen and phosphorus are limiting, usually simultaneously. For chlorophyll, the overall picture is considerably different compared to begin April. In areas where light is limiting, concentrations are generally higher because the solar radiation has increased. In areas where nutrients have become limiting concentrations are significantly lower than begin April. This is true in particular where the distance to the Rhine outflow larger. Thus chlorophyll decreases as the distance from the coast increases. The changes in the pattern of limiting factors also lead to changes in the composition of the phytoplankton. Diatom concentrations have declined, but still this group is present nearly everywhere. *Phaeocystis* is outcompeted in many regions by flagellates due to phosphorus limitation and is now confined to a small range of 20 km immediately off shore. Flagellates are now the most abundant group, though not in the south, where light is the main limiting factor. Here dinoflagellates start to increase.

Autumn situation (Figure 7) During the autumn, light limitation is restricted to more or less the same areas compared to the summer situation. The coverage of areas with nitrogen limitation has slightly increased. There is less phosphate limitation in the coastal zone and no phosphate limitation outside the 50 km zone. The chlorophyll pattern is more or less the same compared to the summer situation, and the levels are of the same magnitude. The species composition has changed drastically. The group of dinoflagellates has become dominant as they have a relatively high growth rate at higher temperatures and are well adapted to nitrogen limitation. Flagellates remain in a small strip before the coast of Holland and the Wadden Sea where there is still phosphate limitation. Diatoms only appear in light limited zones like the plume of the river Rhine.

TREND ANALYSIS

After calibration, the model was used to simulate a 20 year period from 1975 up to 1994. The simulation takes the variability of nutrient discharges, irradiation and water temperature into account. Natural variations and/or possible trends in transport, boundary conditions and suspended matter concentrations were not taken into account. In combination with measurements, the simulation results were analysed to determine the changes in system behaviour over time: the actual trend analysis (de Vries et al., 1997). The simulation can also be regarded as a model validation because none of the model coefficients where changed. To give an impression of the model behaviour Terschelling 4 km offshore was chosen, a location which is relatively sensitive towards changes in (anthropogenic) nutrient discharges by the river Rhine. At this location a small downward trend in both measured and computed chlorophyll concentrations is detected as a result of a decrease in nutrient discharges starting mid 80's. Figure 8 shows slightly increasing concentrations of ortho-phosphate from 1975 up to 1980, a stabilisation at rather high levels, followed by a decrease. Every year ortho-phosphate is depleted. The period in which ortho-phosphate is depleted has become somewhat longer during recent years. Measured and simulated nitrate and silicate agree well. There is no observed or simulated trend. Depletion of both nutrients occurred every summer.

CONCLUSIONS AND FUTURE DEVELOPMENTS

Replacing the DYNAMO model by BLOOM and high resolution transport modelling resulted in a huge step forward in modelling eutrophication in the Dutch coastal zone. The model provides an adequate reproduction of a twenty year period of field observations and reproduces steep gradients perpendicular to the coast. Still further improvement of the biological processes in the model is foreseen in the near future.

Biological processes The model provides a rather reliable reproduction of observed chlorophyll levels. The modelled changes in species composition is however less reliable because of insufficient knowledge of certain (groups of) species. In co-operation with the Dutch Institute for Research of the Sea (NIOZ) competition experiments are being carried out to establish BLOOM coefficients for various algal species, thus improving the capability of the model to compute species composition. During summer, the computed chlorophyll levels are high compared to in-situ measurements. This is likely caused by grazing of zooplankton, a process not included in the model. The model will be extended with dynamical grazing. The modelling of sorption kinetics of phosphate and the sediment-water exchange of nutrients is somewhat simplistic. A more elaborate model of the sediment and inorganic kinetics of phosphate is operational and will be applied in the near future.

Trend analysis The anthropogenic phosphorus discharges into the Dutch part of the North Sea have declined recently. However the impacts on the North Sea ecosystem have so far been limited. Chlorophyll levels, an important target parameter of Dutch policy, are hardly affected. Only in phosphate limited areas, for example near the coast of Terschelling, chlorophyll levels declined slightly. The nitrogen discharges have not declined over the past twenty years. A reduction seems desirable because algal growth is frequently limited due to the depletion of nitrogen.

REFERENCES

Delft Hydraulics (1991). Mathematical simulation of algae blooms by the model BLOOM II. *Report T68.*

Delft Hydraulics (1992). Proces formuleringen DBS. *Report T542 (inDutch).*

Delft Hydraulics (1994). Model validation study DBS in networks. *Report T1210.*

Peeters, J.C.H., H.A. Haas, and L. Peperzak (1991). Eutrofiering, primaire produktie en zuurstofhuishouding in de Noordzee. *National Institute of Coastal and Marine Management. Report GWAO-91.083 (in Dutch)*

Vries, I. de, R.N.M. Duin, J.C.H. Peeters, F.J. Los, M. Bokhorst, R.W.P.M. Laane (1997). Patterns and trends in nutrients and phytoplankton in Dutch coastal waters: comparison of time series analysis, ecological model simulation and mesocosm experiments. *Submitted.*

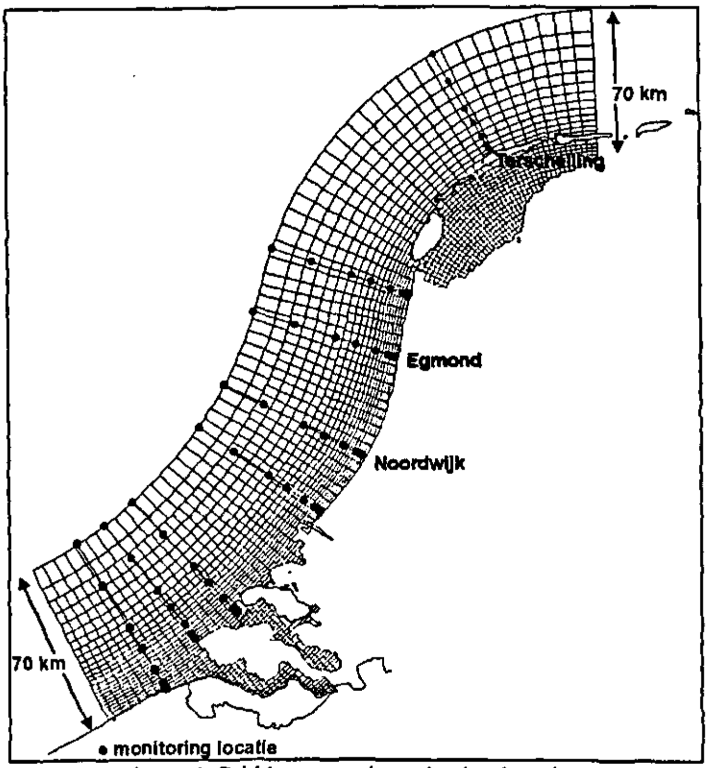


Figure 1 Grid layout and monitoring locations

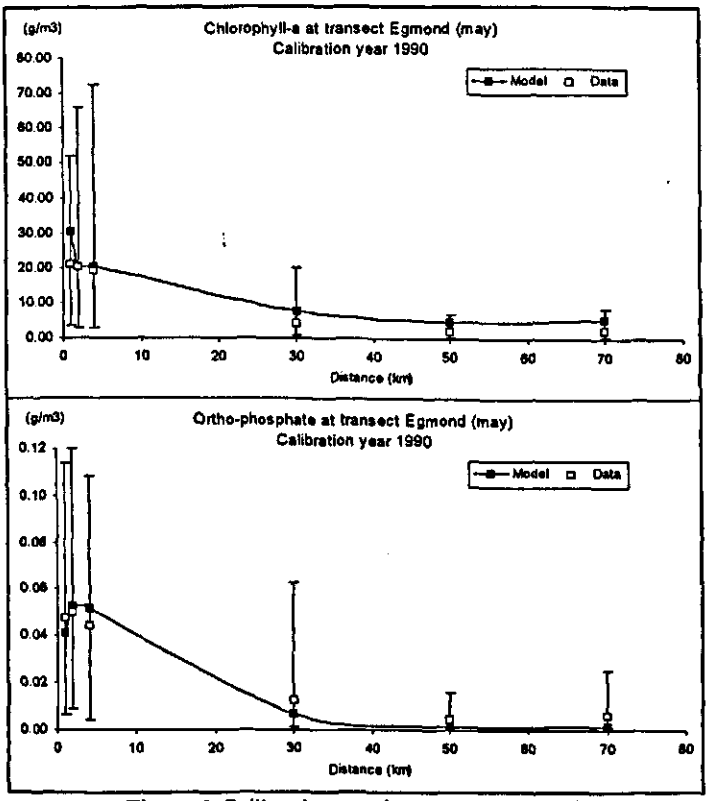


Figure 2 Calibration result transect Egmond.

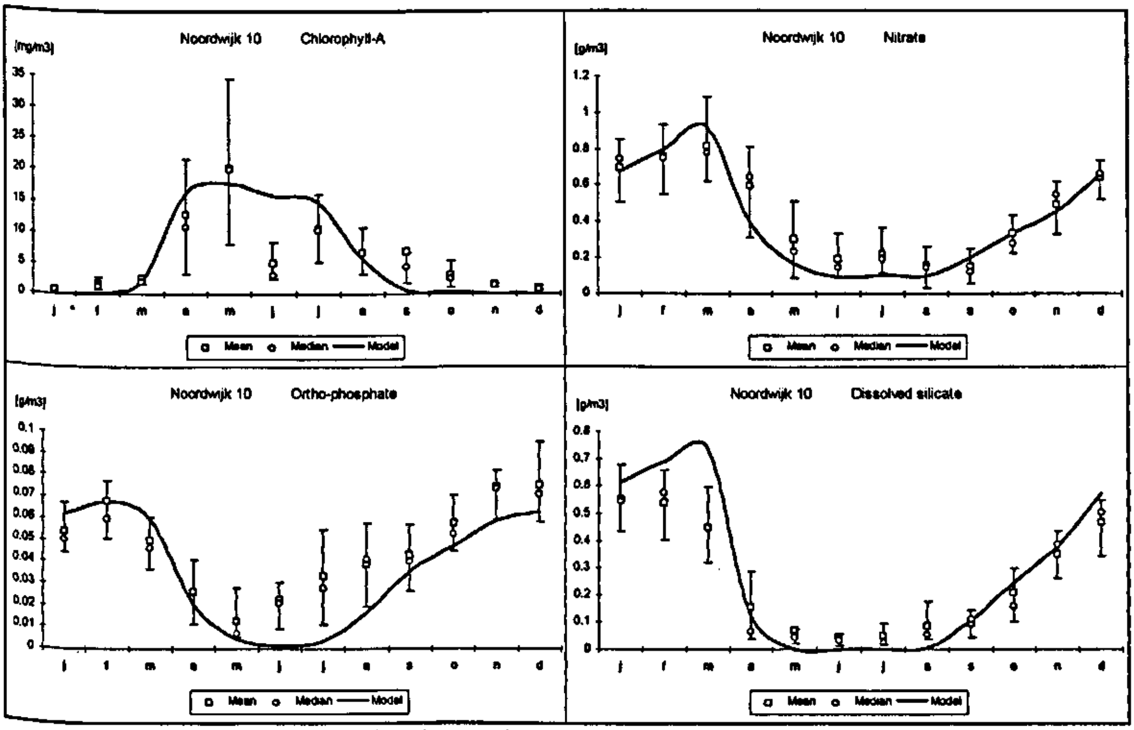


Figure 3 Calibration result station Noordwijk 10 km offshore.

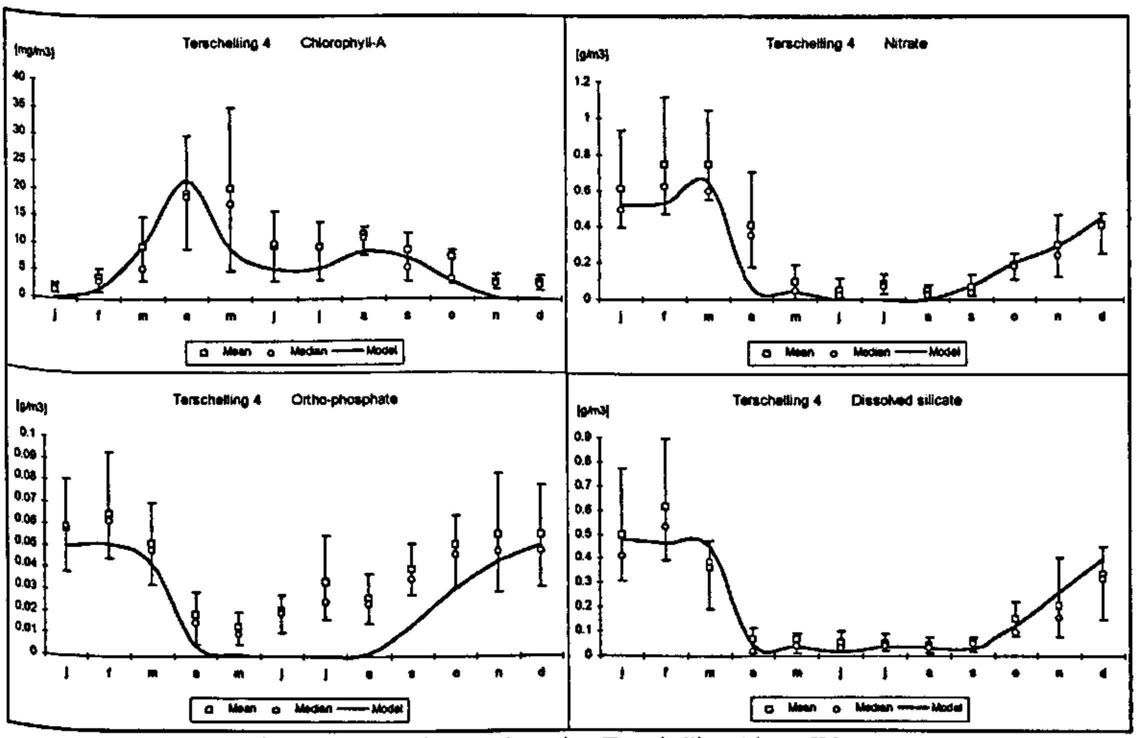


Figure 4 Calibration result station Terschelling 4 km offshore.

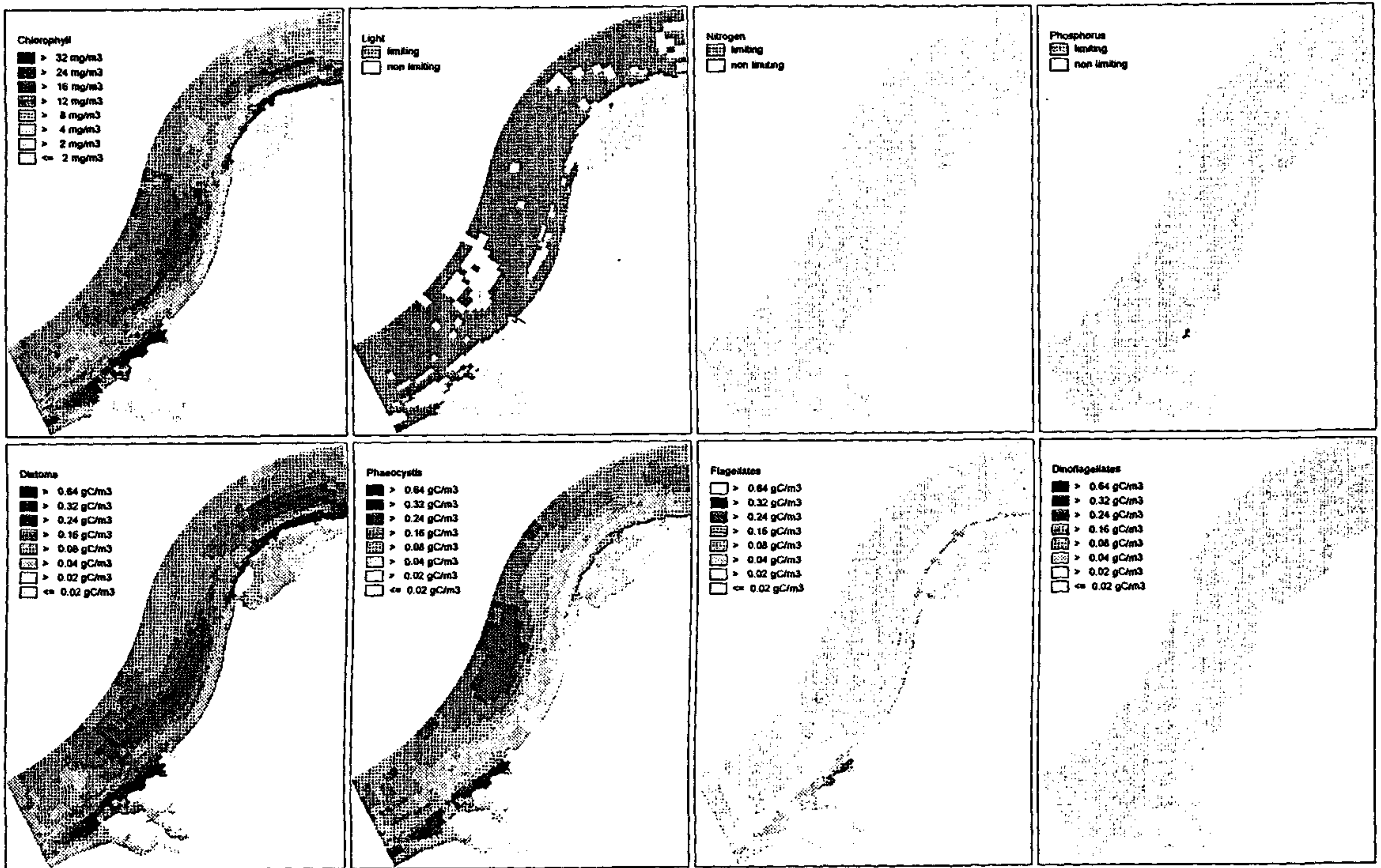


Figure 5 Simulated spatial distribution of chlorophyll-α, limiting factors and species composition begin April.

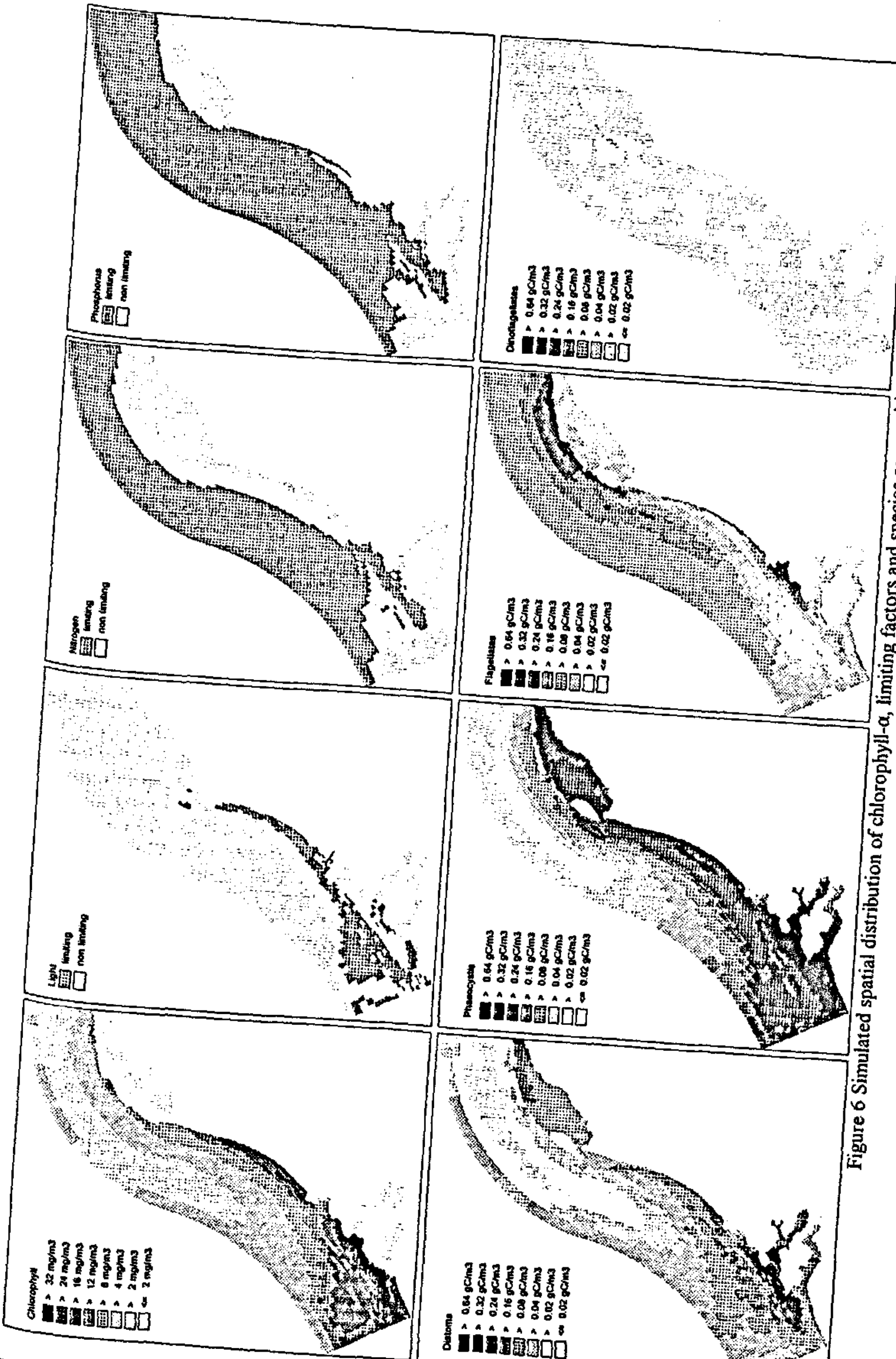


Figure 6 Simulated spatial distribution of chlorophyll-α, limiting factors and species composition begin June

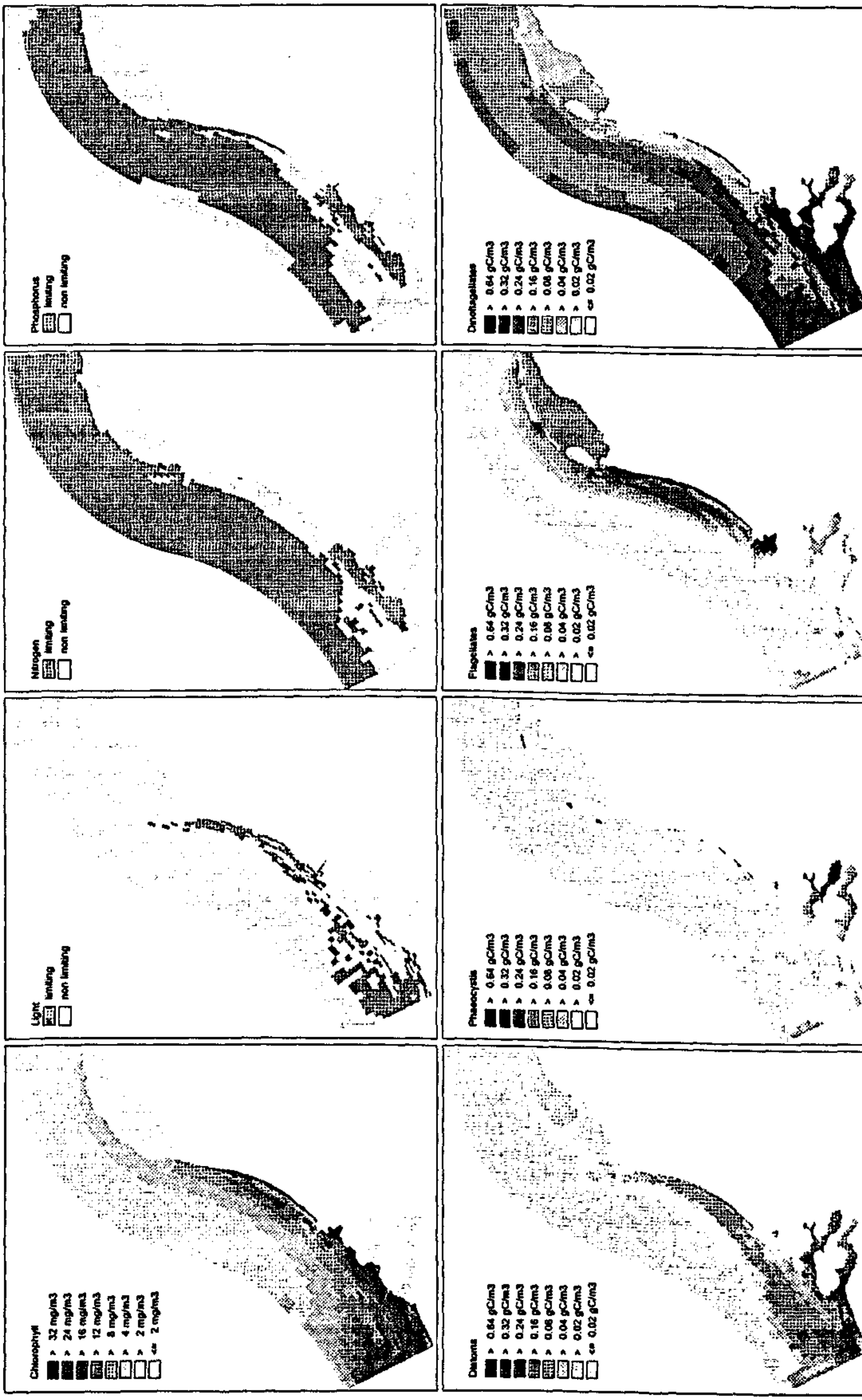


Figure 7 Simulated spatial distribution of chlorophyll-α, limiting factors and species composition begin August.

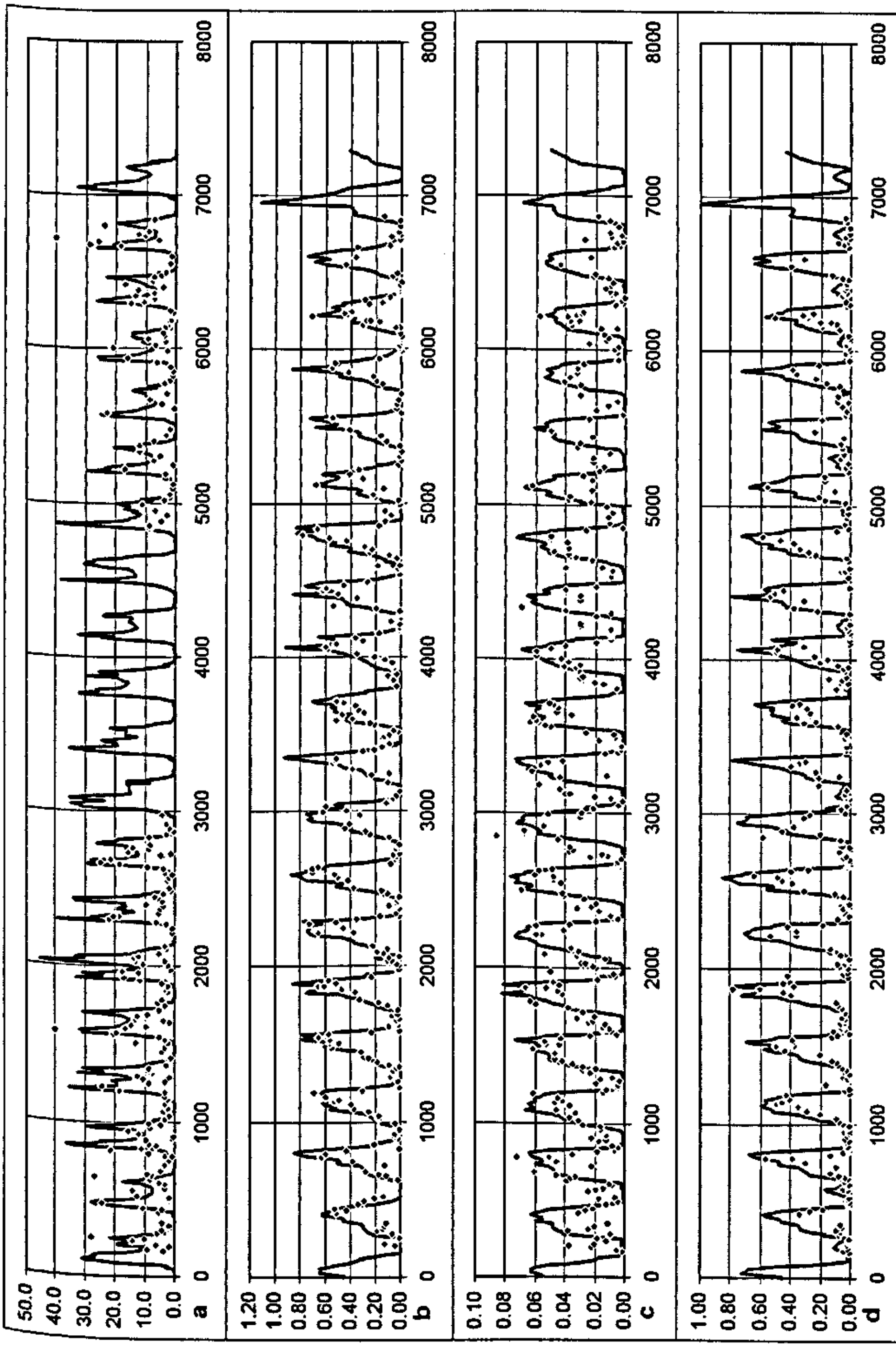


Figure 8 Validation result for station Terschelling 4 km offshore 1975-1994
(a: chlorophyll-α in µg/l, b: nitrate in mg/l, c: ortho-phosphate in mg/l, d: dissolved silicate in mg/l)

A MODEL OF DITCH VEGETATION IN RELATION TO EUTROPHICATION

Jan H. Janse

RIVM, Laboratory for Water and Drinking Water Research, P.O. Box 1, 3720 BA Bilthoven, The Netherlands

ABSTRACT

A functional model of a ditch ecosystem has been developed, aimed at describing the relation between nutrient input and the water quality and dominant vegetation in polder ditches. The model describes the competition between several functional groups of macrophytes: submerged, as well as algae. The groups have been defined according to the layer(s) in which they grow: submerged, floating or emergent, rooted or non-rooted. The model also includes the cycling of nutrients within the water, the upper sediment and the vegetation. The model has been applied to the data of 8 experimental ditches, receiving different levels of nutrient loading during 4 years (Eugelink *et al.*, 1997). The controls and the low- and medium-loaded ditches remained dominated by submerged plants, while in the high-loaded ones a dense cover of duckweeds developed. In the sand ditches, the submerged biomass was lower than in the respective clay ditches. The model simulations by *PCDitch* were generally comparable to these field observations, although the biomass of the submerged groups and the division between them still needs to be improved. A sensitivity analysis revealed that, among others, the maximum growth rates, the minimum phosphorus contents and the P adsorption constant, were important parameters. An optimisation study has been performed for the sensitive parameters, minimising the total sum of squared differences between simulated and measured values for all ditches, leading to the best *overall* fit. The model may be used for the derivation of the 'critical loading level' for a shift from submerged vegetation to duckweed dominance.

KEYWORDS

Ditch; duckweeds; eutrophication; macrophytes; phosphorus; nitrogen; nutrients

INTRODUCTION

Drainage ditches are small, linear water bodies, with as main task the discharge of surplus water from lowland agricultural areas. They form the link between the farmland and larger waters such as lakes and canals. Many ditches also serve to transport water *to* the fields during periods of shortage, mainly in summer. With a total length of about 300,000 km, ditches are a common water type in the Netherlands, mainly in the western and northern parts of the country. They may be found in lowland parts of other countries as well. Besides their hydrological functions, ditches may have an important ecological function, providing a habitat for many plant and animal species. The water depth is typically less than 1.5 m. Because of their shallowness, ditches are often dominated by macrophytes, besides epiphytic, benthic and filamentous algae. Most ditches require yearly maintenance (removal of the vegetation and/or the detrital layer).

Ditches are among the water bodies which are strongly affected by eutrophication (and by pesticides as well). Agricultural run-off is usually the dominant source of nutrients in these waters. Eutrophication generally causes a shift from a submerged vegetation with a vertical growth strategy, to a vegetation with a horizontal growth strategy (Bloemendaal & Roelofs, 1988). The extreme of this is a surface layer of pleustophytic plants, such as duckweeds (*Lemnidae*). While a submerged vegetation coincides with a high oxygen concentration and a rich animal life, a duckweed layer leads to anoxic conditions in the water and loss of animal diversity. In about one-third of the ditches in the Netherlands this is a common phenomenon.

An important question is at what level of nutrient input this shift occurs. To address this question, the mathematical model *PCDitch.* has been developed. The model describes the relation between the external nutrient loading, the nutrient concentrations and the biomass of the dominant vegetation types in ditches: submerged plants, duckweeds and algae. Its aim is to assess the 'critical level' of nutrient loading above which undesired effects, such as a duckweed or algal dominance, are likely to occur. Some other important system characteristics, such as the water depth, hydrology and sediment type, are taken into account, as they codetermine the systems's response to nutrient loading. The model may thus assist in the derivation of loading standards and, together with run-off models, in agricultural scenario analyses. The model is confined to the ditch ecosystem (water column and upper sediment layer) itself; the relation between land use and nutrient run-off is not a part of the model.

The model differs from descriptive-statistical approaches, such as regression analysis of duckweed cover on nutrient concentrations (e.g. De Groot *et al.*, 1987) in the fact that a number of explaining processes has been included, in order to widen its application potential. At the same time, the model differs from structural statistical models (e.g. Barendrecht, 1990; STOWA, 1993) which focus on species composition. These different approaches should be regarded as complementary to each other.

MODEL DESCRIPTION

An overview of the model structure of PCDitch is shown in Figure 1. The model may be regarded as a competition model between several functional groups of water plants, within the context of a general description of the nutrient cycles. The main 'goal variables' of the model are the biomasses of the different plant groups, notably the relative abundance of duckweeds versus submerged plants and the abundance of filamentous algae, as well as the phosphorus, nitrogen and oxygen concentrations. The model includes both the water column and the upper sediment layer, both assumed to be homogeneous and well mixed. The model describes the cycles of four elements: dry weight, phosphorus, nitrogen and oxygen. All biotic components as well as detritus are modelled in multiple units. This is done to achieve closed nutrient cycles within the model system, and to account for the high variability of the nutrient ratios in the field, notably depending on the nutrient loading level.
The components of the model are (modelled in [g m^{-3}] or [g m^{-2}]):

- Inorganic nutrients (PO_4, NH_4, NO_3), both in water and pore water
- Detritus in water
- Sediment detritus
- Oxygen in the water (O_2).
- Total algae
- Water plants, divided in the following functional groups:
    + rooted submerged angiosperms ('Elod')
    + charophytes ('Char')
    + non-rooted submerged angiosperms ('Cera')

+ freely-floating plants (duckweeds) ('Lemn')
+ rooted, floating-leaved plants (Nymphaeids) ('Nymp')
+ emergent plants (helophytes) ('Helo')

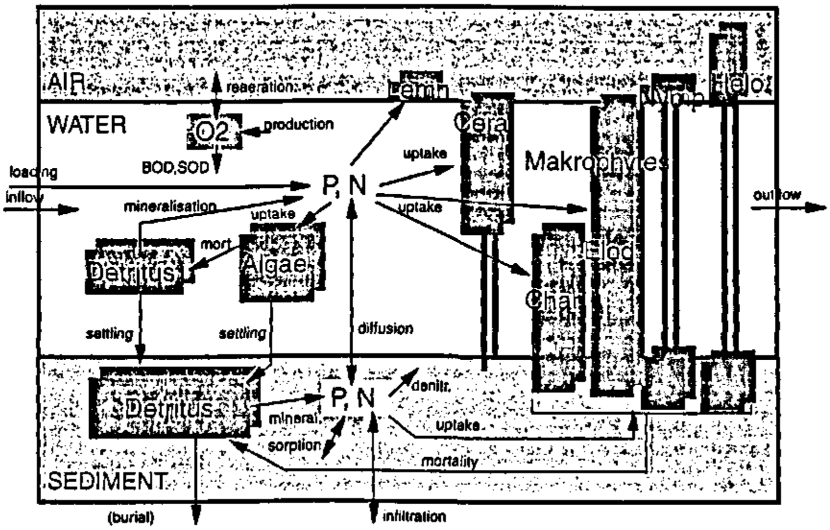


Fig. 1. PCDitch model structure.

The user should define the (initial) thickness of the sediment layer, as well as some sediment parameters such as the density, porosity, lutum content and initial organic matter content. External loading, dilution and infiltration define the in- and outflow of matter. General processes affecting detritus and inorganic nutrients were derived from the model PCLake (Janse and Aldenberg, 1990; Janse, 1996); they include (net) settling, burial and mineralisation of detritus, adsorption (phosphorus only), nitrification and denitrification (nitrogen only), and diffusive transport between water and sediment. Most oxygen proceses have been linked to the organic matter processes. Reaeration is assumed to be hampered by duckweed (e.g. Portielje and Lijklema, 1995).

The biotic components of the model are the different functional groups of water plants mentioned above, and planktonic or filamentous algae, which are lumped in one functional group. Animal groups such as zooplankton, macrofauna and fish have been left out, as they are considered not very important for the prediction of the primary producers. The definition of the plant groups distinguished is primarily based on the layer(s) in which they grow and the layer(s) where they get their nutrients from. The classification into 16 growth forms given by Den Hartog & Segal (1964) and Den Hartog & Van der Velde (1988) has been used as a template. Several groups were lumped, while others were left out because they are not common in ditches. The characteristics of the plant groups are summarised in table 1. The root and shoot fractions of each group are assumed as constant in time. The submerged plants were divided in elodeids+potamids (rooted, growing in the entire water column) and ceratophyllids (non-rooted, canopy-formers). The charophytes (rooted, growing in the lower part of the water column) were distinguished because of their special character as macro-algae. All submerged groups may compete not only for light and nutrients, but also for space. The lemnid group (duckweeds) are the small, floating plants, which take their nutrients from the water column only. They may hamper the growth of the submerged vegetation due to light interception. Nymphaeids may hamper the growth of submerged vegetation via light interception and the growth of duckweeds via competition for space, while helophytes may take considerable amounts of nutrients from the sediment (In the field, however, they are often removed once a year by man). The number and the definition of the plant groups is made flexible. The outcome of the competition between the plant groups is mainly determined, in the model, by the factors light, space, nutrients and - for algae and duckweeds - dilution.

Table 1. Characteristics of the functional groups of macrophytes

Parameter	Elod	Char	Cera	Lemn	Nymp	Helo
emergent part	0	0	0	0	0	0.5
floating part	0	0	0	1.0	0.25	0
submerged part	0.95	0.95	1.0	0	0	0
root fraction	0.05	0.05	0	0	0.75	0.5
distribution of submerged part	water column	lower half	upper half	n.appl.	n.appl.	n.appl.

For each group, the modelled processes are modelled: nutrient uptake, growth, respiration and mortality. Nutrient uptake and growth are described separately (Janse and Aldenberg, 1990). The production is a function of the following factors: maximum growth rate, water temperature, light interception at the water surface, under-water light climate (for submerged plants only), internal P and N contents of the plants (nutrient limitation), and the plant biomass already present in the particular layer (competition for space or other density dependent factors). For duckweeds, a number of parameters have been derived from STOWA (1992). Regarding mortality, a simple phenomenological approach was chosen, following Scheffer *et al.*, 1993) and Wortelboer (1990). It is assumed that the mortality is low in spring and summer, and increased from a certain, predefined date at the start of autumn, causing die-off of the vegetation until a certain surviving fraction is left over which is available again at the start of the next growing season. Mortality may also may include (mechanical) removal for management purposes. For duckweeds only, passive migration has been added as an additional loss factor. The algae have been modelled as in the lake model PCLake (Janse and Aldenberg, 1992).

SITE DESCRIPTION

System characteristics

The artificial ditches, located at Renkum, the Netherlands, have been constructed in 1987-1988; they have been described extensively by Portielje (1994). All ditches are 0.5 m deep in the middle, 1.6 m wide at the bottom and 40 m long and have a gravel bank with a slope of 30°. Four of the ditches have a lightly clayish sediment, the other ones a sandy sediment. All ditches were left undisturbed for one year. Between 1989 and 1992, nutrient treatments took place. Each series consisted of a control and three levels of nutrient addition (denoted as 'low', 'middle' and 'high'). The 'low-loaded' ditches twice yearly received a dose of 0.26 gP m^{-2}. The 'middle-loaded' ditches received 0.82 gP m^{-2} and 3.1 gN m^{-2}. The two highly-loaded ditches received ten doses a year of 1.4 gP m^{-2} and 8.5 gN m^{-2}. Besides these nutrient additions, the ditches received dry and wet atmospheric deposition, estimated as 3.0 10^{-4} [gP m^{-2} d^{-1}] and 0.009 [gN m^{-2} d^{-1}]. Especially the N load from this source is considerable. The additions were chosen as to cause a P concentration of 0.15 mgP l^{-1} (the Dutch standard for lakes) in the low-+treated ditches, 3 times this value in the middle-treated and 25 times in the high-treated ones. The total N loadings were 6 times the P loadings; in the low-treated ditches it was somewhat higher because this ratio was exceeded by the atmospheric load already.

The sediment characteristics and the initial values of the model variables were derived from the measurements given in the table 2 (from Portielje, 1994).

Table 2. Sediment characteristics of the ditches

Variable	Unit	Clay ditches	Sand ditches
Lutum (<2 μm)	[%]	10.3 ± 0.4	1.2 ± 0.1
Porosity (below 1 cm)	[-]	0.4	0.3
ρ (below 1 cm)	[g m^{-3}]	$1.5 \cdot 10^6$	$1.7 \cdot 10^6$
Organic matter	[% of DW]	0.75 (0.7-0.8)	0.13 (0.1-0.2)
Extractable P	[mgP gDW^{-1}]	0.25 (0.22-0.29)	0.10 (0.10-0.11)
Total N	[mgN gDW^{-1}]	0.50 (0.41-0.60)	0.05 (0.04-0.07)
Total Fe	[mgFe gDW^{-1}]	11 (10-12)	1.7 (1.6-1.8)
Oxalate-extractable Fe	[mgFe gDW^{-1}]	3.0	0.07
Extractable Al	[mgAl gDW^{-1}]	0.001	0.00032
$CaCO_3$	[% of DW]	0.4 (0.3-0.6)	< 0.1

We assumed a relevant sediment layer of initially 5 cm. An important difference between the two sediment types is thus that the clay ditches contain 5 times as much organic matter, twice as much phosphorus and 6 times as much nitrogen than the sand ditches. In the clay ditches, the phosphorus is mainly in organic form and adsorbed on iron, while in the sand ditches aluminium is the main adsorbent.

Summary of field data

The vegetation succession in the ditches has been described by Portielje (1994) and Eugelink and Lijklema (1997). In short, the charophytes in the clay ditches gradually disappeared, the fastest the higher the loading. They were replaced by elodeid plants, the density of which was a function of the loading level. In the middle-loaded ditch they declined again in 1994. Occasionally, blooms of filamentous or planktonic algae were observed. Also helophytes developed in some of the ditches. The highly-loaded ditch was dominated by duckweed from 1991 on. In later years, the duckweed declined dramatically in summer due to grazing by lepidopteran larvae. In the untreated sand ditch no plant life was seen at all apart from some benthic algae. The low-treated ditch also showed benthic algae, together with some elodeids from 1993 on. The middle-treated ditch was dominated by algae (benthic, filamentous and planktonic) with some elodeids from 1995 on, while the highly-loaded was dominated by duckweeds, like the respective clay ditch.

SIMULATION METHOD

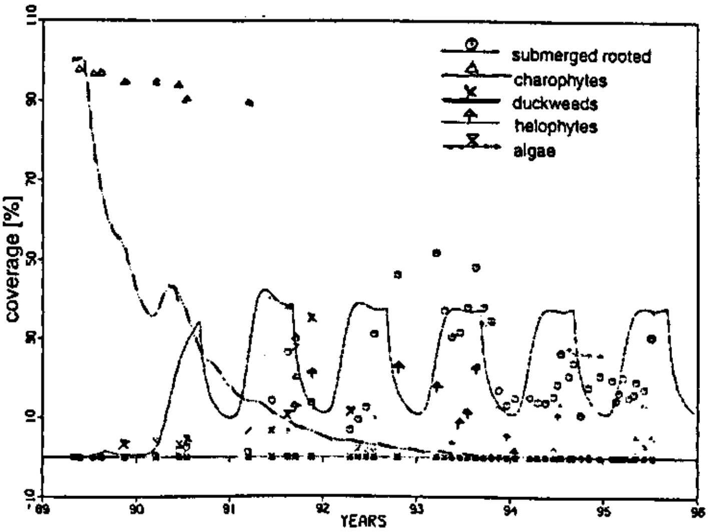
The simulations started in May 1989, with measured densities, i.e. a charophytes vegetation in the clay ditches and no macrophytes present in the sand ditches. It seemed most reasonable to start with these densities as initial values, as processes likely to affect initial colonisation of a new water body, such as migration and germination ecology, were not modelled. Inorganic matter in the water column was arbitrarily set at 1.0 g m^{-3} and the initial SRP concentration in the water at 0.01 mgP l^{-1}, while the sediment parameters were set as in table 2. All process parameters were the same for all ditches. A number of sensitive parameters (listed in table 3) were selected for an optimisation study, calibrating on the vegetation monitoring data (coverage percentages) of all 8 ditches. The duckweed data of the high-loaded clay ditch were corrected as if no herbivory by lepidopteran larvae had taken place, as these larvae were not included in the model. As yet, also the algal data were left out because of incomplete data. The fit criterion to be minimised (maximum likelihood) was the sum of squared differences between model and data for all data points in all ditches for all vegetation groups. In this way, the method seeks for that parameter combination that gives the best *overall* fit regarding all ditches, rather than the best fit for one ditch while leaving a poor fit for the others.

Table 3. Parameters used in the optimisation study

Parameter	Unit	Description	Range	Value
μMaxElod	$[d^{-1}]$	max. growth rate of submerged rooted plants	0.2 - 0.5 (0.8)	0.29
μMaxChar	$[d^{-1}]$	max. growth rate of charophytes	0.2 - 0.5	0.20
μMaxCera	$[d^{-1}]$	max. growth rate of non-rooted submerged plants	0.2 - 0.4	0.40
μMaxLemn	$[d^{-1}]$	max. growth rate of duckweeds	0.2 - 0.4	0.29
μMaxNymp	$[d^{-1}]$	max. growth rate of nymphaeids	0.03 - 0.10	0.033
μMaxHelo	$[d^{-1}]$	max. growth rate of helophytes	0.03 - 0.10	0.043
μMaxPhyt	$[d^{-1}]$	max. growth rate of algae	1.0 - 2.0	1.90
hLRefElod	$[W\ m^{-2}]$	Monod constant for light	20 - 50	31.6
hLRefChar	$[W\ m^{-2}]$	Monod constant for light	10 - 50	18.6
hLRefCera	$[W\ m^{-2}]$	Monod onstant for light	20 - 50	39.1
hLRefPhyt	$[W\ m^{-2}]$	Monod constant for light	10 - 30	10.2
cPDElodMin	$[mgP\ mgD^{-1}]$	min. P content	0.0005 - 0.002	0.0008
cPDCharMin	$[mgP\ mgD^{-1}]$	min. P content	0.0005 - 0.002	0.0012
cPDCeraMin	$[mgP\ mgD^{-1}]$	min. P content	0.0005 - 0.002	0.0012
cPDLemnMin	$[mgP\ mgD^{-1}]$	min. P content	0.004 - 0.006	0.0041
cPDPhytMin	$[mgP\ mgD^{-1}]$	min. P content	0.001 - 0.005	0.0020
fHiberElod	[-]	overwintering fraction	0.2 - 0.8	0.21
fHiberChar	[-]	overwintering fraction	0.6 - 1.0	0.91
fHiberCera	[-]	overwintering fraction	0.2 - 0.8	0.71
fHiberLemn	[-]	overwintering fraction	0.1 - 0.4	0.13
fHiberHelo	[-]	overwintering fraction	0.2 - 0.5	0.43
kdPAdsIMS	$[m^3\ gD^{-1}]$	affinity constant for P adsorption	0.001 - 0.015	0.0074

RESULTS

Fig. 2 shows the results using the calibrated values for the parameters in table 3, and the default values for all others. (Only the control and high-loaded clay ditches are shown). In general, the dominance of submerged vegetation in the control, low- and medium-loaded ditches, and the shift towards duckweed dominance after two years in the heavily-loaded ones, is modelled reasonably well. The same holds true for the disappearance of the charophytes, although the rate is overestimated for the control ditch. The vegetation densities in the sand ditches are much lower than those in the clay ditches, which is in accordance with the monitoring data. The algal densities are generally underestimated, as well as the helophytes in the control ditch. The absence of non-rooted plants and nymphaeids is simulated well.



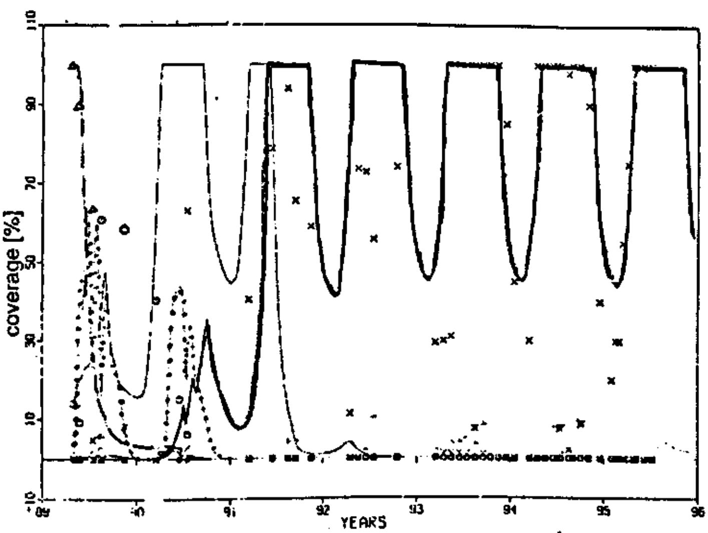


Fig. 2. Simulation and monitoring data for the control (upper graph) and high-loaded (lower graph) clay ditches. Treatment period: May 1989 - April 1993. See legend in upper graph; non-rooted submerged plants and nymphaeids were not present. Plant densities are expressed in coverage percentages.

DISCUSSION AND CONCLUSIONS

The model reproduces correctly a dominance of submerged vegetation at phosphorus input concentrations up till 0.45 mgP l^{-1}, and a rather rapid shift to duckweed dominance at an input concentration of 3.5 mgP l^{-1}. The simulated submerged biomass (or coverage) is related to the nutrient loading level (up till 0.45 mgP l^{-1}) and to the sediment type (clay or sand). The simulation of the competition between submerged plants and algae, however, which is equally important in view of water quality and ecological values, remains poor. This may be partly due to the fact that the algal data were not yet apt for use for the calibration. Continuing of the calibration on more data, also on the nutrient concentrations, using a broader set of parameters, is likely to further improve the model fit. This will facilitate interpolation of the results to intermediate loading levels.

The aim of the model, to describe the relation between nutrient run-off, water quality and vegetation in ditches in a general way, makes extrapolation of the results to field situations necessary. The model application on the artificial ditches as described here is likely to be useful for this purpose, because both the input and the response variables are rather well defined. In field situations, however, a number of additional intervening variables are to be accounted for. These include, among others, the water depth and water level fluctuations, the hydrology (water retention time, infiltration, seepage), and vegetation management. A field validation will be necessary to be able to derive the 'critical' loading levels for undesired ecological effects in different situations. In view of the remaining uncertainty in the model structure and parameters after calibration, these loading levels will have to be defined in a probabilistic way.

ACKNOWLEDGEMENTS

The provision of data by the Agricultural University (prof.dr. L. Lijklema, ir A.H. Eugelink) and the Winand Staring Centre (ir. J. Drent), both at Wageningen, is gratefully acknowledged.

REFERENCES

Aarts, E.H.L. & P.J.M. van Laarhoven (1989). Simulated Annealing: an introduction. Stat. Neerl. 43 (1): 31-52.

Bloemendaal, F.H.J.L. & J.G.M. Roelofs, red. (1988). Waterplanten en waterkwaliteit. KNNV, Utrecht.

Eugelink, A.H., R.M.M. Roijackers & L. Lijklema (1997). Responses of vegetation on nutrient loading in shallow aquatic ecosystems. Submitted.

Groot, W.T. de, F.M.W. de Jong & M.M.H.E. van den Berg (1987). Arch. Hydrobiol. 109: 601-618.

Hartog, C. den & S. Segal (1964). A new classification of the water-plant communities. Acta Bot. Neerl. 13: 367-393.

Hartog, C. den & G. van der Velde (1988). Structural aspects of aquatic plant communities. In; Symoens, J.J. (red.). Vegetation of inland waters. Handbook of Vegetation Science, 113-153.

Janse, J.H. (1997). A model of nutrient dynamics in shallow lakes in relation to multiple stable states. Hydrobiologia 342/343: 1-8.

Janse, J.H. & T. Aldenberg (1990). Modelling phosphorus fluxes in the hypertrophic Loosdrecht lakes. Hydrobiol. Bull. 24: 69-89.

Portielje, R. (1994). Response of shallow aquatic ecosystems to different nutrient loading levels. Diss., LU Wageningen.

Portielje, R. & L. Lijklema (1995). The effect of reaeration and benthic algae on the oxygen balance of an artificial ditch. Ecol. Mod. 79: 35-48.

Scheffer, M., A.H. Bakema & F.G. Wortelboer (1993). Megaplant, a simulation model for aquatic macrophyte dynamics. Aquat. Bot.

STOWA (1992). Ontstaan en bestrijden van deklagen van kroos. 1. Literatuur. 2. Modelmatige benadering van de kroosontwikkeling en beoordeling van beheersbaatheid. Report no. 92-09 and 92-10 (in Dutch).

STOWA (1993). Ecologische beoordeling en beheer van oppervlaktewater. Beoordelingssysteem voor sloten op basis van macrofyten, macrofauna en epifytische diatomeeën, en Wetenschappelijke verantwoording van het beoordelingssysteem voor sloten. STOWA-rapporten 93-14 en 93-15.

Traas, T.P. & T. Aldenberg (1997). A mass-balance for the logistic population growth equation and implications for biomass turnover. In prep.

Van der Molen, D. & A. Breeuwsma (1995). Nutriëntenverliezen in de landbouw; kloof tussen onvermijdbare en acceptabele nutriëntenverliezen? Landschap 12 (6): 7-14 (in Dutch).

Wortelboer, F.G. (1990). A model on the competition between two macrophyte species in acidifying shallow soft-water lakes in the Netherlands. Hydrobiol. Bull. 24: 91-107.

Modelling the competition between buoyant cyanobacteria, green algae and dinoflagellates to assist the implementation of artificial mixing in lakes and reservoirs

*[x]B.W. Ibelings, *W.M. Wiegant, [#]R.H. Aalderink, [†]H.A.M. Ketelaars, [‡]G.W.A.M. Waajen, [§]J. Koedood.

*HASKONING - Nijmegen; [#]Wageningen Agricultural University, Department of Water Quality Management and Aquatic Ecology; [†]Water Storage Corporation Brabantse Biesbosch Ltd. - Werkendam.; [‡]Water Authority West-Brabant - Breda; [§]Water Management and Sewerage Department - Hilversum; [x]Present address of corresponding author: National Institute of Inland Water Management and Waste Water Treatment - P.O. Box 17 - 8200 AA - Lelystad - The Netherlands.

ABSTRACT

Blooms of nuisance cyanobacteria are still a major problem facing water management in The Netherlands. In some lakes (e.g. in the deep lake Nieuwe Meer) blooms were successfully prevented by artificial mixing. In a fully mixed lake cyanobacteria like *Microcystis* no longer benefit from their buoyancy and are furthermore unable to acclimate to the highly variable light regime or suffer negative consequences of a decreased pH. Artificial mixing, however, is not without risks: nutrients formerly locked into the unproductive hypolimnion become available for phytoplankton growth. Results with the technique are variable. We feel there is a need to study the application of artificial mixing carefully for each system considered. Two models were developed to study the growth of cyanobacteria and their non-buoyant competitors (in these models green algae) in artificially mixed systems. Results from the models showed that: i) artificial mixing must cover the whole lake, untreated areas enhance the risk of *Microcystis* blooms; ii) the mixing velocity must be adequate to entrain buoyant cyanobacteria; iii) successful control of algal blooms requires a sufficient decrease in light availability, hence a sufficient depth of the system; iv) the mixing regime - permanent vs. intermittent mixing - is crucial for success; v) artificial mixing of deep lakes also reduces the risk of nuisance blooms in shallow, connected water systems.

KEYWORDS

Artificial mixing, buoyancy, *Ceratium*, cyanobacteria, dinoflagellates, green algae, *Microcystis*, modelling, *Oscillatoria*, waterblooms, water management.

INTRODUCTION

Surface blooms of cyanobacteria are perhaps the most striking visible exponent of freshwater eutrophication. Reynolds & Walsby (1975) formulated three conditions for the formation of surface waterblooms: i) an abundant cyanobacterial population; ii) buoyancy; iii) stability of the water column. Buoyancy is generated by gas-vesicles inside the cell, that lower the overall density of the cyanobacteria below the density of water (Walsby, 1994). Stability of the water column is the result of surface heating by the sun; the resulting gradient in density provides resistance against wind-mixing. In a stable water column buoyant *Microcystis* colonies float up into the shallow near surface mixed layer, where the increased light dose may enhance growth of the cyanobacteria (Ibelings *et al.*, 1991). In a water column lacking stability wind induced mixing easily exceeds the floating velocity of cyanobacteria (Reynolds, 1987), hence the colonies are no longer able to position themselves in the water column: growth is arrested (Reynolds *et al.*, 1983), and accumulation at the lake surface is prohibited. Since stability of the water column is vital to both enhanced growth of cyanobacteria and to the process of surface scum formation, artificial mixing of lakes has been used successfully to reduce problems with nuisance cyanobacteria. In the Nieuwe Meer, The Netherlands, a relative abundance of *Microcystis* of more than 90 % was reduced to less than 10 % after introduction of artificial mixing (Visser *et al.*, 1996a).

Artificial mixing may be a promising measure to prevent blooms of nuisance algae, it is, however, not always successful. As a consequence of artificial mixing the total planktonic biomass may even increase (Pastorok *et al.*, 1981). Destratification releases nutrients, that were formerly locked into the deep, unproductive hypolimnion, and renders them available for phytoplankton growth. A personal survey of the literature shows that in nearly fifty percent of all cases phytoplankton biomass (mg Chl*a* m^{-2}) increased, a.o. in the Nieuwe Meer (Visser *et al.*, 1996a). An increase in phytoplankton can be prevented, however, if mixing is sufficiently deep to impose a light limitation on phytoplankton growth. Artificial mixing still was considered successful in the Nieuwe Meer, since *Microcystis* was replaced by green algae. Furthermore the *phytoplankton concentration* (mg Chl*a* m^{-3}) decreased due to the increase in mixing depth, and transparency increased. In the Biesbosch reservoirs, The Netherlands, on the other hand blooms of *Microcystis* occur regularly, despite artificial mixing (Oskam & van Breemen, 1992), hence the therapy can not always prevent growth of nuisance algae.

Two models were developed, that assist the implementation of artificial mixing. The models were used to answer the following main questions that remained unanswered from a managers point of view:
- does artificial mixing affect nutrient levels in the lake?;
- does artificial mixing affect total Chl*a* levels in the lake?;
- does artificial mixing affect the ratio cyanobacteria : green algae?

In the paper we will show results from the two models that were applied to a number of lakes and drinking water reservoirs in The Netherlands: Biesbosch reservoirs, Lake de Kuil, and Lake Sloterplas (see table 1).

Table 1. General characteristics of the water systems discussed in the paper; Nieuwe Meer, where artificial mixing has been applied succesfully is given for comparison.

Variable	Units	Biesbosch reservoirs			Sloterplas	De Kuil	Nieuwe Meer
		De Gijster	Honderd en Dertig	Petrusplaat			
Surface area	ha	305	210	100	78	11	132
Volume	10^6 m^3	40	33	13	16	0.8	24
Maximum depth	m	27	27	15	30	10.5	30
Mean depth	m	13	15	13	20	7.3	18

The problems in Lake De Kuil differ from the other two systems. In the Biesbosch reservoirs and Lake Sloterplas *Microcystis* spp. Are the dominant cyanobacteria. In Lake De Kuil (used for swimming) nuisance surface blooms of the red cyanobacterium *Oscillatoria rubescens* occur occasionally when the filaments leave their position on the metalimnion. Destratification would remove the metalimnion, where the population of *Oscillatoria* builds up; this is not well tolerated by the cyanobacterium (Reynolds, 1984). In Lake De Kuil the dinoflagellate *Ceratium hirundinella* is one of the dominant phytoplankton species. In other systems artificial mixing has resulted in massive blooms of *Ceratium*, even resulting in fishkills (Nichols *et al*, 1980). In contrast to cyanobacteria, *Ceratium* does not float but swims, using flagella. Like cyanobacteria it can benefit from its vertical migration to improve its light climate: it actively seeks optimal irradiances (Heaney & Talling, 1980). A priori it seems likely that mixing velocities that exceed the swimming velocity of would suppress growth, and reduce the risk of a dinoflagellate bloom. Hence artificial mixing is potentially a good measure against these dinoflagellate blooms as well. In addition to modelling competition between buoyant cyanobacteria and green algae in all three systems, competition of green algae with the free swimming dinoflagellate *Ceartium hirundinella* was also tested for Lake de Kuil.

METHODS

In the study two different models were used to describe growth of buoyant cyanobacteria and their non-buoyant algal competitors. Next we will give a short description of the two models, showing their strengths and weaknesses. The models were applied to investigate the potential of artificial mixing for controlling algal blooms in the three aforementioned systems.

Model Y. This model was developed in DUFLOW, a micro-computer package for simulation of flow and water quality in networks of open water courses. The model includes the cycling of nutrients and describes growth of two algal species, to incorporate competition between cyanobacteria and green algae (see table 2 fopr characteristics). Sediment-water exchange is modelled dynamically, using the concept of an active sediment top-layer of fixed depth. For a detailed description of the processes refer to van Duin *et al.* (1995). The water system is subdivided into a number of nodal points and sections to represent the water system geometry. For this study the water quality model was extended to enable simulation of (de)stratified systems. In the vertical direction the water column is split into two well mixed layers, representing the epi- and hypolimnion. Mixing between the two layers is described by an exchange of flow. The depth of the thermocline is fixed, assuming a stable stratified system. Algal growth is restricted to the epilimnion using a depth averaged expression for light limitation, i.e. the model does not take vertical migration of cyanobacteria into account (this aspect of cyanobacterial growth is covered by model Z). Buoyancy in model Y was simulated only by a lower sedimentation loss rate of the cyanobacteria (see table 2).

Table 2. Some characteristics of green algae and cyanobacteria, as used in model Y.

Variable	Units	Green algae	Cyanobacteria
Chl*a* : Carbon	$\mu g\ mg^{-1}$	30	15
optimal irradiance for growth	μmol photons $m^{-2}\ s^{-1}$	60	15
maximal growth rate	day^{-1}	1.5	0.5
loss rate	day^{-1}	0.15	0.02
sedimentation velocity	$m\ day^{-1}$	0.5	0.01
Monod constant for P	$mg\ P\ l^{-1}$	0.005	0.005

Model Z. Model Z is an elaboration of the model published by Kromkamp & Walsby (1990), that describes the vertical migration of cyanobacteria - as a consequence of changes in buoyancy - in a non-turbulent water column. Changes in buoyancy are the result of storage and consumption of ballast carbohydrates, produced by photosynthesis. The main additions to the existing model are the impact of turbulent mixing (both induced artificially and by wind) on the position of the cells, competition for the available resources (light and nutrients) between cyanobacteria and green algae, and growth of the species. The light dose received by the phytoplankton is determined by i) the daylength, ii) surface irradiance, iii) extinction of light under water, and iv) the position of the cells in the water column, which is dependent on a) wind induced mixing, b) vertical migration by cyanobacteria, and c) artificial mixing. The description of nutrient dynamics, and description of the water systems geometry is, however, less accurate than in model Y.

The movement of algae, caused by Eddy-dispersion, is described as a random-walk (Brown & Rothery, 1993), in which the windspeed determines the stepsize, as described by Henderson Sellers (1984). Several solutions were tested to describe the impact of artificial mixing on water movement. Eventually artificial mixing was described as suggested by Goossens (1979) - basically an increase in turbulence of the water column. In model Z 300 cyanobacterial colonies, and 300 green algal cells receive a diameter - via a randomgenerator - based on a log-normal distribution, derived from field-measurements on *Microcystis* and *Scenedesmus* (Ibelings *et al.*, 1991). The model tracks the movement, position and growth of each individual particle, to which a variable biomass is coupled, and via this mass a Chl*a* concentration, determined by the particle's growth and loss rates. The time step for calculation was 6 minutes.

Slight modifications of the model were required to study vertical migration by *Ceratium hirundinella*. *Ceratium* migrates to light intensities between 120 - 150 µmol photons m^{-2} s^{-1}, with a swimming velocity that can be varied between 0.001 - 0.003 m s^{-1} (Gissel Nielsen, 1991).

RESULTS AND DISCUSSION

In this paper we discuss the impact of artificial mixing on water quality and the development of phytoplankton. Our main aim in using the models is to compare stratified vs. destratified conditions in lakes; we did not strive to reach a perfect reproduction of measured concentrations of Chl*a*, P-PO_4 etc., although concentrations produced by the models were certainly acceptable.

General results of model Y - impact of destratification on water quality of lakes. Destratification of the lakes in this study will increase the availability of P-PO_4, as was shown by calculations using model Y (figure 1). The increase in concentrations of dissolved nutrients is partially the result of a reduction in the Chl*a* concentration - especially in the Sloterplas (figure 2), but also because nutrients formerly locked into the hypolimnion now become available. The reduction in Chl*a* in the Sloterplas is the result of a stringent light limitation caused by the increase in mixing depth. The average depth of Lake De Kuil is only just sufficient to prevent an increase in the cyanobacterial biomass in response to the increased availability of P-PO_4 (figure 1); a reduction is not achieved by artificial mixing (figure 2).

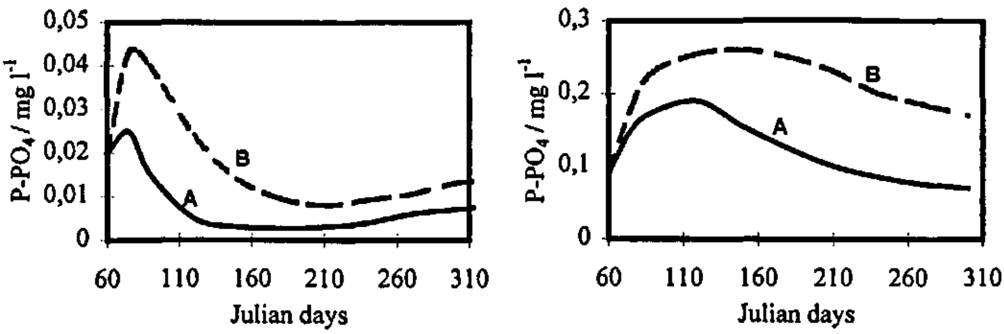
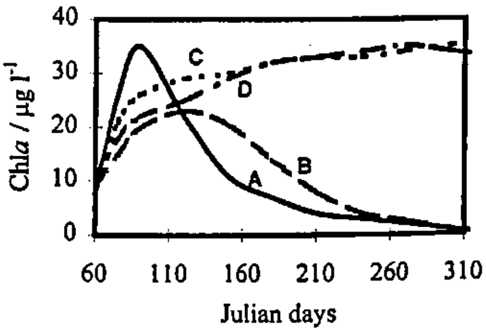


Figure 1. Concentration of P-PO_4 in Lakes De Kuil (left figure) and Sloterplas (right figure) in the epilimnion under stratified conditions (A) and under destratified (B) conditions. Results model Y.

Development of phytoplankton in shallow canals connected with the Sloterplas. Blooms of *Microcystis* occur regularly in this deep lake. Nuisance blooms of cyanobacteria also occur in an extensive network of shallow canals, situated in residential areas, that are connected with the lake. The following question was addressed using model Y: If growth of *Microcystis* in the lake is restricted by artificial mixing of the deep Sloterplas, will the nuisance surface blooms in the shallow water systems also disappear?

The results of model Y show that mixing the Sloterplas does result in a nearly complete removal of cyanobacteria from the connected shallow canals (figure 3). Green algae still reach high concentrations, mixing of the Sloterplas only results in a delay of the green algal development in these canals. These results are especially sensitive to the value of the dispersion coefficient, D, that determines the degree of exchange between the lake and the canals in the model. At realistic values of D of up to 5 m^2 s^{-1}, however, the results are as shown in figure 3.



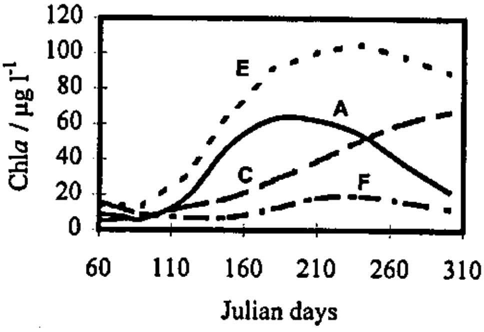


Figure 2. Concentration of Chl*a* of green algae and cyanobacteria in Lakes De Kuil (left figure) and Sloterplas (right figure)) in the epilimnion under stratified conditions and under destratified conditions. Results model Y: A = green algae/stratified; B = green algae/destratified; C = cyanobacteria/stratified; D = cyanobacteria/destratified; E = total Chl*a*/stratified; F = total Chl*a*/destratified.

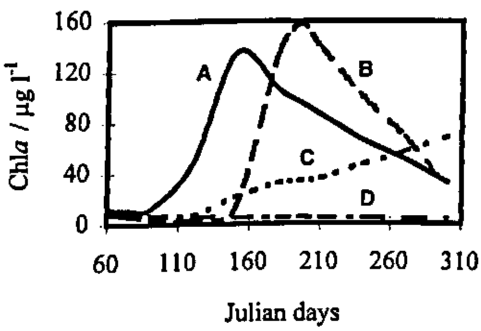


Figure 3. Chl*a* concentration of green algae and cyanobacteria in shallow canals connected with Lake Sloterplas. The figure shows the impact of artificial mixing of the deep Sloterplas on the development of algae in the shallow canals. Results model Y: A = green algae/stratified Sloterplas; B = green algae/destratified Sloterplas; C = cyanobacteria/stratified; D = cyanobacteria/destratified.

General results of model Z - behaviour of buoyant cyanobacteria. The following results of model Z (not shown) are in good agreement with general ecological knowledge of *Microcystis* (e.g. Reynolds, 1987; Ibelings *et al.*, 1991; Walsby, 1994), and strengthen our confidence in the model:

- buoyant cyanobacteria gain a competitive advantage over green algae in deep, partial stable systems;
- in a stable non-mixed water column *Microcystis* performs a diel vertical migration, as was shown earlier in the model of Kromkamp & Walsby (1990). The maximum depth of migration is determined by the light extinction coefficient of the water column;
- at higher wind speeds the competitive advantage of buoyancy decreases. Growth rate of *Microcystis* at a wind speed of 9 m s^{-1} was only 15-30 % of the growth rate at a wind speed of 0.5 m s^{-1};
- artificial mixing, like wind-mixing, reduces growth and production of buoyant cyanobacteria, especially at higher mixing speeds;
- moderate mixing favours larger colonies (which have a higher floating velocity) over smaller colonies;
- mixing results in a decrease of the ratio cyanobacteria : green algae.

Explanations for the reduction of cyanobacteria through artificial lake mixing. All phenomena given below may contribute to the decline of cyanobacteria and the shift towards other phytoplankton, that is often observed after implementation of artificial mixing:

- cyanobacteria are superior competitors for inorganic carbon at high pH; artificial mixing however, reduces the pH of lakes. Additionally, at low pH lysis of cyanobacteria through the activity of cyanophages increases (Shapiro, 1984);

- buoyant, colony-forming cyanobacteria depend on (partial) stability of the water column in their competition for light with other non-buoyant phytoplankton (Ibelings *et al*, 1991);
- artificial mixing results in a more dynamic underwater light climate; cyanobacteria acclimate less well to these conditions than eukaryotic algae (Ibelings *et al.*, 1994);
- sedimentation losses of non-buoyant competitors of cyanobacteria, especially green algae, decrease in a well mixed lake (Visser *et al.*, 1996a).

Application of model Z to the Biesbosch-reservoirs. The Biesbosch system consists of three interconnected reservoirs. Water from the River Meuse is pumped into the first reservoir, De Gijster; from De Gijster water is pumped into reservoir Honderd en Dertig and eventually into reservoir Petrusplaat (see table 1). All reservoirs are deep and mixed artificially, but an extensive shallow area of 6 m depth - where artificial mixing is absent - extends over the first 100 metres of De Gijster. Questions addressed were:
- is this shallow area responsible for blooms of *Microcystis* that occur in De Gijster?;
- is the presence of *Microcystis* in the other two reservoirs, which do not have shallow areas the result of autonomous growth, or is growth initiated by the direct connection with De Gijster?

Visser *et al.* (1996b) investigated whether water column stability of the unmixed relatively shallow area in De Gijster exceeded water column stability in the deep, main part of the reservoir. Limited evidence for this was found in an enhanced buoyancy loss of *Microcystis* in the shallow area, that could not have occurred in a fully mixed watercolumn. Model Z was used to compare growth of *Microcystis* in a 15 m deep, completely mixed reservoir, in a 6 m deep non-mixed reservoir, and in a reservoir consisting of 60 % deep/mixed and 40 % shallow/non-mixed. The results are shown in figure 4. Growth of *Microcystis* in the latter type reservoir starts earlier, and the eventual biomass is higher than in the 15 m deep reservoir, hence growth of Microcystis is stimulated by the presence of the shallow, unmixed area of De Gijster.

Does the enhanced growth of *Microcystis* in De Gijster have an effect on the development of *Microcystis* in reservoir Honderd en Dertig? Figure 4 also shows the impact of the connection of Honderd en Dertig with De Gijster: At a low initial concentration of *Microcystis* in reservoir Honderd en Dertig, growth of *Microcystis* starts earlier in the season, which leads to a higher population density than in the scenario where Honderd en Dertig remains isolated from De Gijster. For a typical K-strategist like *Microcystis* the size of the inoculum, and the timing of the onset of growth are of vital importance (Reynolds, 1987). A reduced inoculum, or delayed growth results in lower population densities.

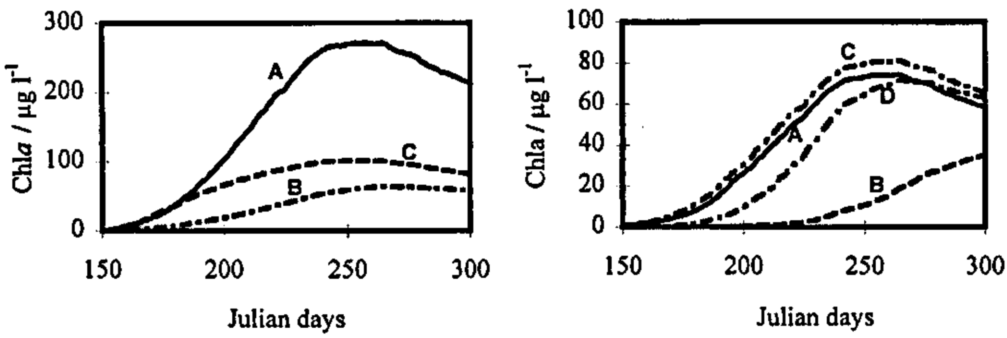


Figure 4. Left figure: development of *Microcystis* in reservoir De Gijster, assuming: depth of 6 m and no artificial mixing (A); depth of 15 m and artificial mixing (B); 40 % surface area of 6m deep without artificial mixing and 60 % surface area 15m deep with artificial mixing and exchange rates bwteen the two parts of 2 volume % per day (C). Right figure: Development of *Microcystis* in reservoirs De Gijster and Honderd en Dertig assuming an isolated reservoir Honderd en Dertig (A and B), or inflow from reservoir De Gijster (C and D), and an initial Chl*a* concentration - inoculum - in reservoir Honderd en Dertig of either 1.0 (A and C) or 0.02 µg Chl*a* l^{-1} (B and D). Results model Z.

The importance of mixing intensity and -regime. Reduction of nuisance algae through artificial mixing requires an adequate mixing velocity and mixing regime. Forsberg & Shapiro (1980) found in enclosure studies that slow mixing rates benefited cyanobacteria, whereas faster mixing rates reduced cyanobacteria. Mixing in the Nieuwe Meer has been designed to exceed the mean floating velocity of *Microcystis*: this ensures that cyanobacteria are entrained in the turbulent flow, and no longer benefit from their buoyancy. Intermittent mixing was superior to permanent mixing in controlling blooms of *Oscillatoria agardhii* in the Fischkaltersee, because acclimation of cyanobacteria to changing environmental conditions is more reluctant than that of eukaryotic algae (Steinberg & Zimmermann, 1988; Ibelings *et al.*, 1994). The importance of mixing velocity and mixing regime was tested - using model Z - especially for Lake De Kuil, where not only cyanobacteria but also the dinoflagellate *Ceratium hirundinella* had to be controlled. The model clearly indicates that growth of *Ceratium* decreases with an increase in the value of the mixing coefficient (figure 5). This figure also indicates that permanent mixing is more efficient in suppressing *Ceratium* than intermittent mixing, whereas the highest population densities are achieved if no artificial mixing is present.

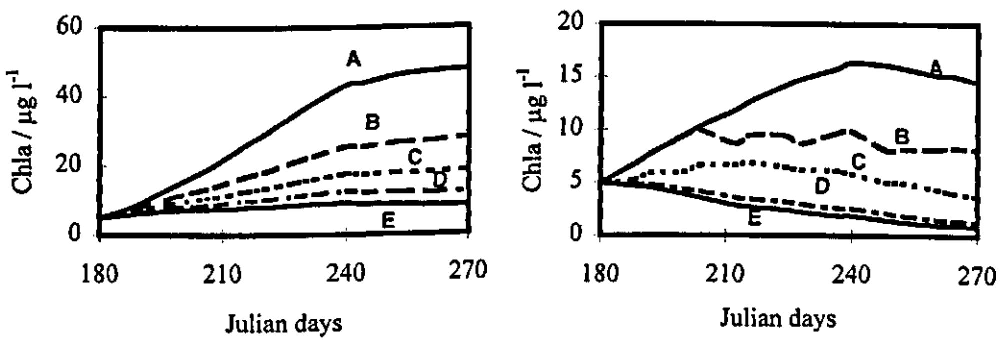


Figure 5. Development of *Ceratium hirundinella* under various artificial mixing conditions in Lake De Kuil; the figure shows the impact of mixing intensity and mixing regime. Left figure: variable mixing coefficients ($m^2 s^{-1}$) of 0 (A); 0.00005 (B); 0.00010 (C); 0.00015 (D); 0.00020 (E). Right figure variable mixing regimes: no artificial mixing (A); artificial mixing only if *Ceratium* > 10 µg Chl*a* l^{-1} (B); mixing only at low wind speed / high insolation (C); wind powered mixing apparatus - mixing only if wind speed > 2 m s^{-1} (D); permanent mixing (E). Results model Z.

CONCLUSIONS

(i) Installation of artificial mixing as a therapy measure against nuisance algae needs careful consideration. An increase in the availability of nutrients is not unlikely, hence algal biomass may increase. (ii) In most instances, however, artificial mixing does result in a decrease in the ratio of cyanobacteria : green algae. (iii) Overall depth is an important factor: in a relatively shallow lake like De Kuil the average depth is insufficient to achieve a reduction in algal biomass through light limited growth. (iv) Modelling growth of *Microcystis* in De Gijster showed that although the average depth is sufficiently deep the large, relatively shallow (but still 6 m deep), and unmixed area is responsible for growth of cyanobacteria. (v) The results of the Sloterplas on the other hand, indicate that in a water system consisting of a deep lake and extensive shallow canals (only 1- 1.5 m deep), mixing of the lake removes *Microcystis* from the whole water system. This is in accordance with the ecology of this cyanobacterium, which occurrence is mainly restricted to systems deeper than 4 m (Schreurs, 1992). (vi) Mixing velocity and mixing regime are also important for success of the treatment. The mixing velocity must be sufficient to entrain buoyant cyanobacteria (or swimming dinoflagellates). (vii) Permanent mixing in the models resulted in the most efficient reduction of nuisance algae - which otherwise take advantage of periods of quiescence. Other studies, however, (e.g. Steinberg & Zimmermann, 1988) have shown that in the long term intermittent mixing may be more efficient in controlling cyanobacteria. (viii) Modelling the outcome of the implementation of artificial mixing may assist lake managers in making the right decisions.

LITERATURE

Brown, D. & Rothery, P. (1993). *Models in biology - Mathematics, statistics and computing.* Wiley & Sons, London.

Duin, E.H.S, van, Portielje, R. & Aalderink, R.H., 1995. Modelling flow and water quality in River Vecht using DUFLOW. In: C. Heatwole (ed.). *Water Quality Modelling.* Proceedings of the international conference on water quality modelling. ASAE. Orlando, 313-324.

Forsberg, B.R. & Shapiro, J. (1980). Predicting the algal response to destratification. In: *Restoration of lakes and inland waters.* EPA440/5-81-010. U.S. E.P.A. Washington D.C.

Gissel Nielsen, T. (1991). Contribution of zooplankton grazing to the decline of a *Ceratium hirundinella* bloom. *Limnol. Oceanogr.* **36**, 1091-1106.

Goossens, L. (1979). *Reservoir destratification with bubble columns.* PhD thesis University of Delft.

Heaney, S.I. & Talling, JF. (1980). Dynamic aspects of dinoflagellate distribution patterns in a small productive lake. *J. Ecol.* **68**, 75-94.

Henderson Sellers, B. (1984). *Engineering limnology.* Pitman, Boston, London.

Ibelings, B.W., Kroon, B.M.A., & Mur, L.R. (1994). Acclimation of photosystem II in a eukaryotic green alga and a cyanobacterium to high and fluctuating photosynthetic photon flux densities, simulating those induced by mixing in lakes. *New Phyt.* **128**, 407-424.

Ibelings, B.W., Mur, L.R., & Walsby, A.E. (1991). Diurnal changes in buoyancy and vertical distribution in populations of *Microcystis* in two shallow lakes. *J. Plankt. Res.* **13**, 419-436.

Kromkamp, J., & Walsby, A.E. (1990). A computer model of buoyancy and vertical migration in cyanobacteria. *J. Plankt. Res.* **12**, 161-183.

Nichols, K.H., Kennedey, W. & Hammett, C. (1980). A fish-kill in Heart Lake, Ontario, associated with the collapse of a massive population of *Ceratium hirundinella* (Dinophyceae). *Freshw. Biol.* **10**, 553-561.

Oskam, G., & van Breemen, L.W.C.A. (1992). Management of Biesbosch reservoirs for quality control with special reference to eutrophication. In: Sutcliffe, D.W. & Jones, J.G. (eds.). *Eutrophication: research and application to water supply.* FBA, Ambleside, 197-213.

Pastorok, R., Lorenzen, M.W. & Ginn, T.C. (1981). *Artificial aeration and oxygenation of reservoirs: a review of theory, techniques and experiences.* Final Report TC-3400 for Waterways Experiment Station, U.S. Army Corps of Engineers.

Reynolds, C.S., Wiseman, S.W., Godfrey, B.M. & Butterwick, C. (1983). Some effects of artificial mixing on the dynamics of phytoplankton populations in large limnetic enclosures. *J. Plankt. Res.* **5**, 203-232.

Reynolds, C.S. (1984). Phytoplankton periodicity: the interactions of form, function and environmental variability. *Freshw Biol.* **14**, 111-142.

Reynolds, C.S. (1987). The response of phytoplankton communities to changing lake environments. *Schweiz. Z. Hydrobiol.* **49**, 221-235.

Reynolds, C.S., & Walsby, A.E. (1975). Waterblooms. *Biol. Rev.* **50**, 437-481.

Schreurs, H. (1992). *Cyanobacterial dominance. Relations to eutrophication and lake morphology.* PhD Thesis University of Amsterdam.

Shapiro, J. (1984). Blue-green dominance in lakes: the role and management significance of pH and CO_2. *Int. Revue ges. Hydrobiol.* **69**, 765-780.

Steinberg, C.E.W. & Zimmermann, G.M. (1992). Intermittent destratification: a therapy measure against cyanobacteria in lakes. *Environm. Technol. Lett.* **9**, 337-350.

Visser, P.M., Ibelings, B.W., van der Veer, B., Koedood, J. & Mur, L.R. (1996a). Artificial mixing prevents nuisance blooms of the cyanobacterium *Microcystis* in Lake Nieuwe Meer, The Netherlands. *Freshw. Biol.* **36**, 4350 450.

Visser, P.M., Ketelaars, H.A.M., van Breemen, L.W.C.A. & Mur, L.R. (1996b). Diurnal buoyancy changes of *Microcystis* in an artificially mixed storage reservoir. Hydrobiologia **331**, 131-141.

Walsby, A.E. (1994). Gas vesicles. *Microbiol. Rev,* **58**, 94-144.

MANAGEMENT AND RESTORATION

STABLE STATES, BUFFERS AND SWITCHES: AN ECOSYSTEM APPROACH TO THE RESTORATION AND MANAGEMENT OF SHALLOW LAKES IN THE NETHERLANDS

S. Harry Hosper

Institute for Inland Water Management and Waste Water Treatment (RIZA), P.O. Box 17, 8200 AA Lelystad, the Netherlands. h.hosper@riza.rws.minvenw.nl

ABSTRACT

Lake restoration in the Netherlands has been focused on the control of external P loading from point sources. However, this approach did not result in the water quality desired. The algae-dominated turbid water state may be extremely stable, and then additional measures are necessary to remove certain 'blockages' such as: the persistent bloom of *Oscillatoria* algae, the bloom-mediated P release from the lake sediments, and the abundance of fish, preventing zooplankton and submerged macrophytes from developing . This paper addresses: (1) the need for an ecosystem approach, (2) the resistance of shallow lake ecosystems to changes in nutrient loading, (3) the concept of *stable states, buffers and switches*, (4) the elaboration of this concept for the restoration process of lake Veluwemeer, and (5) the perspectives for lake restoration in the Netherlands. Priority should be given to fighting the *Oscillatoria* blooms. Winter flushing with water low in TP and algae proved to be an efficient tool in reducing these blooms. Winter fishing for planktivorous fish, such as bream and roach, could enhance the top-down control of algae through the grazing by zooplankton, particularly by the large *Daphnia* species.

KEYWORDS

Biomanipulation, eutrophication, lake restoration, *Oscillatoria*, shallow lakes, Veluwemeer

INTRODUCTION

Lake restoration is one of the major issues in water management in the Netherlands. Hundreds of millions of guilders are being spent yearly for nutrient control and additional lake restoration measures (Anonymous, 1995). In the first half of this century, the majority of shallow lakes was clear and the lake bottom was covered with vegetation. Now, most lakes look murky and green because of an excessive algal growth and the resuspension of sediments by fish and wind. Wildlife and recreational values have been severely negatively affected. Submerged plants have disappeared and the fish stock is characterized by large numbers of only a few species of prey fish, like bream and roach, and relatively low numbers of predatory fish, such as pike, pike-perch and perch. Eutrophication, i.e. increased nutrient loading, is the main cause for the deterioration of the lake ecosystems. Other perturbations, such as chemical pollution toxic to zooplankton, and the loss of lake-marginal wetlands, will have reinforced the effects of nutrient enrichment. The lake restoration strategy in the Netherlands has been focused on reduction of the external phosphorus loading. However, so far the control of external loading has not resulted in the water quality desired (Van Liere and Gulati, 1992; De Deckere *et al.*, 1996). The turbid lake ecosystem tends to be resistant to recovery and solely reducing of external nutrient loading seemed to be insufficient for attaining clear water conditions. A comprehensive approach to lake ecosystem functioning (an ecosystem approach) may provide additional tools for lake restoration. The lakes referred to here are shallow (mean depth 1-4 m) and vertically mixed throughout the year.

WHY AN ECOSYSTEM APPROACH?

The development of the lake restoration strategy started with the 'Vollenweider approach', which aimed at criteria for external nutrient loading from the analysis of large numbers of different lakes (multi-lake studies) (Vollenweider, 1968). It then became clear that lake sediments, where most of the phosphorus (P) had accumulated, could act as a source of P, once the external loading had been reduced (Golterman, 1977). So, the reduction in external loading may be counteracted by (enhanced) internal loading, giving a marginal response in water quality. Additionally, ecologists argued that the biotic structure of the turbid water ecosystem (e.g. large numbers of zooplankton-eating and benthos-eating fish, the absence of submerged macrophytes) also contributes to the resistance of the lake to recovery (Shapiro, 1980; Moss, 1983). Shapiro *et al.* (1975) had already emphasized the necessity of treating lakes as ecosystems, rather than 'containers of algae and phosphorus'. For understanding the algal dynamics and the mechanisms causing resistance in lake recovery, an ecosystem approach is needed. Fig. 1 shows a simple model of phytoplankton biomass, as a result of production and loss processes. Production by photosynthesis is controlled by external and internal nutrient loading and the availability of sunlight. Loss processes are consumption by filter-feeders (such as zooplankton and mussels), mortality (due to factors such as limiting resources or parasites), and sinking of algae to the sediments. Inflow and outflow are omitted in this model.

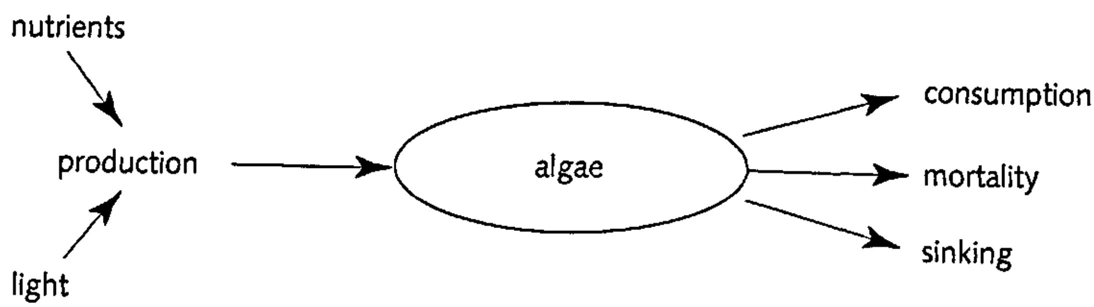


Fig. 1. Algal biomass is the net result of production and loss processes.

The algae are part of the lake ecosystem and the feedback from this ecosystem to the algae is lacking in the above model. Algae affect the physical (e.g. water turbidity), chemical (e.g. pH and oxygen levels) and biological lake conditions (e.g. food web structure, abundance of submerged macrophytes) and this effect is dependent on both the species composition and biomass of the algae. The changed lake conditions may, in turn, promote the algae, leading to a self-perpetuating process of increasing algal blooms (Fig. 2).

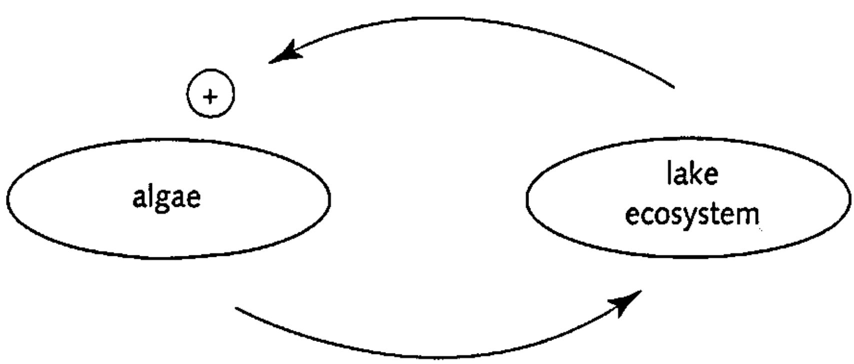


Fig. 2. Interaction of algae and the physical, chemical and biological conditions of the lake ecosystem.

Finally, light availability, which depends on lake depth (or mixing depth for stratified lakes) and non-algal turbidity, determines the upper limit of algal biomass. Understanding these feedback mechanisms is of primary importance for restoration and management of shallow lakes, as can be illustrated by the following examples:

Physical conditions
Algae make the water turbid. Beyond a certain threshold of lake turbidity, populations of submerged macrophytes in shallow lakes collapse, and with that the 'clearing effects' of the submerged vegetation. Providing refuge to grazing zooplankton, competition for nutrients between macrophytes and algae, and protecting the sediments against wind- or fish-induced resuspension, are among the major mechanisms by which vegetation makes the water clearer.

Chemical conditions
Algal production leads to a high sediment oxygen demand (by decomposition of sinking algae) and high pH (by net consumption of CO_2). Both factors, low redox potential and high pH at the water-sediment interface, enhance P release from the sediments (internal loading), thus fueling algal growth.

Biological conditions
Highly productive lake systems favor the development of blue-green algae (cyanobacteria). Cyanobacteria may show conspicuously low rates of loss, due to low maintenance energy requirements (low mortality rates), reduced edibility for zooplankton (low consumption rates), and buoyancy control by gas vacuoles (low sinking rates). Turbid lakes, devoid of vegetation, are low in piscivorous (fish-eating) fish, and high in planktivorous (zooplankton-eating) and benthivorous (benthos-eating) fish. Abundant planktivores control the zooplankton, resulting in low grazing of algae. Large numbers of benthivores, unhindered by plants, stir up the mud, thus contributing to turbidity and P release.

In addition to nutrient enrichment, several other perturbations affecting the lake ecosystem may indirectly result in a further deterioration of lake water quality (Fig. 3). Examples are the loss of shallow lake-marginal wetlands, intentional destruction of aquatic vegetation, heavy stocking of benthivorous fish, the inlet into lakes of external water with a significantly different macro-ionic composition, or pollution with toxic chemicals. The loss of lake-marginal wetlands, by strictly stabilizing the water levels (Coops, 1996) or by drainage, can enhance algal blooms. Wetland areas provide spawning and nursery habitat for the predatory fish pike, thus contributing to top-down control of algal biomass (Klinge *et al.*, 1995). Furthermore, vegetated areas act as a sink for nutrients, fine resuspended sediments and algae. High stocks of benthivores, like common carp and bream, may destroy existing stands of submerged vegetation. Altering the macro-chemistry by letting in river water (rich in macro-ions such as bicarbonate, sulfate and chloride) to peat lakes which were previously isolated, may enhance the internal loading of P from the sediments (Roelofs and Smolders, 1993; Beltman and Van der Krift, 1997). Pollution with chemicals toxic to zooplankton, may result in reduced grazing of algae (Hurlbert, 1975; Scholten *et al.*, 1994). In the 1970s, the Rhine river was acutely toxic to *Daphnia magna* (Sloof *et al.*, 1985), making it most likely that toxic chemicals reinforced the effects of eutrophication in Rhine-fed lakes at that time.

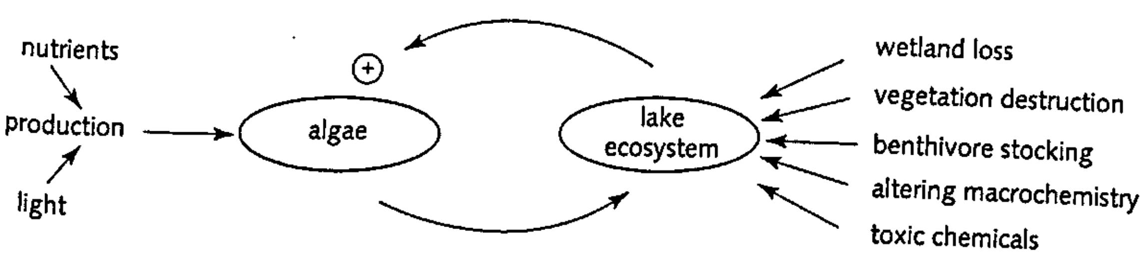


Fig. 3. Perturbations other than increased nutrient loading may also indirectly promote algal blooms.

Eutrophication, i.e. the enrichment with plant nutrients (N, P), is the primary cause of algal blooms and concomitant changes in the lake ecosystem, such as the loss of submerged macrophytes and predatory fish. Lake restoration focuses on the control of P loading, rather than of N loading. This P-oriented approach was taken for practical reasons: (1) according to the limiting nutrient concept, the reduction of only one nutrient is sufficient to control algal biomass; (2) N loading originates to a relatively large extent from non-point sources and therefore is more difficult to control, and (3) some species of cyanobacteria (e.g. *Aphanizomenon, Anabaena*) are able to assimilate dissolved, atmospheric N_2. Furthermore, it can be concluded from case studies that a reduction of P loading, with an unchanged high N loading, has been effective in reducing algal biomass (Hosper, 1997). Unfortunately, case studies of the opposite (a reduction of N loading and high P loading), could not be found.

The relationships between nutrient loading and shallow lake response are complex and tend to be different for the process of eutrophication and the reverse process of oligotrophication. Well-documented, long-term case studies of eutrophication and oligotrophication in shallow lakes are scarce. The description of both processes, given below, is based on 'circumstantial evidence' from case studies in the Netherlands (Hofstra and Van Liere, 1992; Van Vierssen *et al.*, 1994; Noordhuis, 1996; Hosper, 1997) and conceptual modeling (Moss, 1990; Moss *et al.* 1996; Jeppesen *et al.*, 1990b, 1991; Scheffer *et al.*, 1993; Van Vierssen *et al.*, 1994; Hosper, 1997). The bottom line is that both clear water lakes and turbid water lakes are resistant to changes in nutrient loading and show the phenomenon of hysteresis. Only at certain critical nutrient levels (thresholds), does the system show dramatic changes in algal biomass (Fig. 4).

Starting from a macrophyte-dominated clear water state, the process of enrichment can be described as a series of stages and events:

(1) Increase in external nutrient loading, but constant low concentrations in the water.
Loss processes, related to the abundance of macrophytes (reduced resuspension, uptake by plants, denitrification) and the adsorption capacity of the sediments, compensate for the increased loading (not shown in Fig. 4). It is assumed here that P, rather than N, limited algal production in the original, macrophyte-dominated lakes. However, biomanipulation case studies have shown that dense macrophyte beds, which recolonized the lake bed after improvement in the water clarity, tend to reduce TN levels (Van Donk *et al.*, 1993; Meijer *et al.*, 1994). These results suggest that N could have been important as limiting nutrient for algal production, as well.

(2) Increase in TP levels in the water, but constant low algal biomass.
The loss processes can no longer cope with the increased loading and TP levels increase. In certain peat lakes, internal P loading from the sediments may have increased, due to the inlet of external water, which is rich in macro-ions such as bicarbonate, sulfate and chloride (Smolders and Roelofs, 1995; Beltman and Van der Krift, 1997). However, the 'clearing effects' of submerged macrophytes (refuge to algae-grazing *Daphnia* against fish predation, allelopathic effects etc.) still keep the algal biomass at low levels.

(3) Non-linear shift to the algae-dominated turbid water state.
Increasing nutrient levels make the system more susceptible to a collapse of the clear water state. Macrophytes become overgrown with periphyton (Van Vierssen *et al.*, 1994; Van Donk and Gulati, 1995) and blooms of (inedible) phytoplankton species will occur more often (Van Donk and Gulati, 1995), decreasing the chances for submerged vegetation. As most shallow lakes are rather uniform in depth, light conditions are fairly similar throughout the lake. Consequently, with deteriorating light conditions, the macrophytes may suddenly disappear from large areas. Additionally, increased nutrient levels may coincide with increased levels of pollution, including chemicals toxic to *Daphnia* and macrophytes. In the 1970s, the heavily polluted Rhine river water (feeding many lakes) was acutely toxic to *Daphnia magna* (Slooff *et al.*, 1985). Other perturbations, such as the intentional destruction of submerged vegetation, heavy benthivore stocking or exceptional weather conditions, may also trigger the shift to the turbid water state (see next section). The TP thresholds are difficult to specify. It is tentatively concluded from a large amount of field data in Denmark on TP, Secchi depth and vegetation (Jeppesen *et al.*, 1990), that the clear water state may be stable up to TP = 100 mg m^{-3} (Hosper, 1997). In small lakes (< 3 ha), however, a sustainable clear water state is possible at higher TP levels, due to the relatively strong impact of submerged vegetation in small water bodies.

(4) Increase in algal biomass, roughly following the P-limitation line.
The slope of the line may differ from lake to lake, due to variation in the CHL/TP ratios for different phytoplankton species and in the availability of TP for algal biomass.

(5) Algal biomass is leveling off.
The upper limit is determined by the availability of light, which depends on mixing depth (lake depth) and non-algal turbidity. In shallow lakes in the Netherlands, the final phytoplankton community consists of either filamentous and well-mixed cyanobacteria (e.g. *Oscillatoria agardhii*) in the shallower lakes (z = 1-2 m) or a mix of green algae and diatoms, with occasional blooms of colonial and scum-forming cyanobacteria (e.g. *Microcystis aeruginosa*) in the deeper lakes (z = 2-4 m) (Berger, 1987; Schreurs, 1992; Ibelings, 1992; Mur and Schreurs, 1995). TP may continue to rise, due to bloom-mediated P release from the sediments (Hosper, 1997).

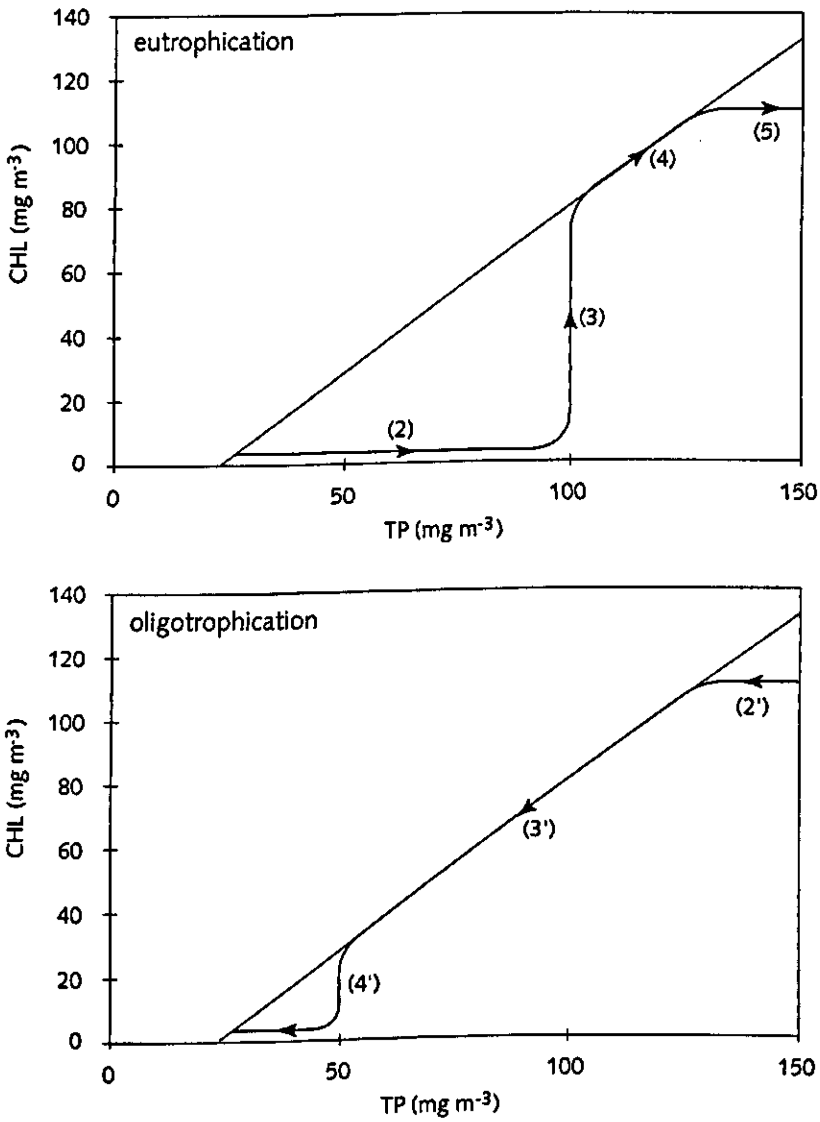


Fig. 4. Effects of eutrophication (upper panel) and oligotrophication (lower panel) on summer algal biomass for a hypothetical shallow lake, showing the phenomenon of hysteresis. The line indicates maximum chlorophyll *a* in relation to TP (Hosper, 1997).
See text for explanation.

The reverse process of oligotrophication runs a different course in the lower TP ranges. Starting from an algae-dominated turbid water state:

(1') Decrease in external P loading, but only a minor reduction in summer TP.
Summer TP levels may be controlled to a large extent by the strong internal P loading from the sediments (not shown in Fig. 4).

(2') Decrease in TP levels, but constant high summer algal biomass.
P is in excess and algal biomass is limited by other factors, e.g. light or N.

(3') Decrease in algal biomass, roughly following the P-limitation line.
The phytoplankton becomes P-limited and algal biomass goes down. The threshold TP level where the phytoplankton shifts from light limitation to P limitation depends on lake depth and non-algal turbidity. As stated above, the slope of the P-limitation line may vary with phytoplankton species and other factors. Secchi depth shows a gradual increase or may jump to higher values, with a shift in phytoplankton species (e.g. from *Oscillatoria* to green algae).

(4') Shift to the macrophyte-dominated clear water state.
If TP reaches such a low level that the transparency of the water allows the re-establishment of macrophytes, the lake (or the shallower parts of the lake) shifts to the clear water state. This lower TP threshold depends on phytoplankton species composition and lake depth. *Oscillatoria*-dominated lakes may produce more algal biomass per unit P than lakes with a mixed phytoplankton community and consequently the TP threshold will be lower for these lakes. It will be clear that macrophyte re-establishment in deeper lakes requires a lower algal biomass (and lower TP) in the water column. Apart from the impact of non-algal turbidity, clear water (SD > 1 m) in shallow lakes can be expected at TP < 50 mg m^{-3} or even lower levels for *Oscillatoria*-dominated lakes (Hosper, 1997). Strong wind-induced resuspension of sediments may prevent the lake from clearing up.

Note that in a certain TP range, both turbid water and clear water may be possible as alternative stable states. Ideally, lake restoration efforts should be aimed at TP < 50 mg m^{-3}. However, in the TP range of about 50-100 mg m^{-3} (or higher TP levels for very small lakes), additional in-lake measures or special natural events may trigger the shift from the algae-dominated turbid water state to the macrophyte-dominated clear water state (see next section).

THE CONCEPT OF *STABLE STATES, BUFFERS AND SWITCHES*

In shallow lakes, phytoplankton blooms, and particularly blooms of *Oscillatoria agardhii*, may be self-reinforcing and therefore resistant to the reduction of external P loading. Additionally, resuspension of sediments by wind and benthivorous fish may contribute to the stability of the turbid water state. Conversely, in shallow lakes of moderate productivity, the submerged vegetation plays a key role in stabilizing the clear water state. TABLE 1 summarizes the buffering mechanisms for the turbid water state and clear water state, as well as the switches (special natural events or actions) which may trigger a shift from the one state to the other.

TABLE 1 STABLE STATES, BUFFERING MECHANISMS MAINTAINING THE STABLE STATES AND SWITCHES TO TRIGGER A SHIFT FROM A STABLE TURBID WATER STATE TO A STABLE CLEAR WATER STATE (FORWARD SWITCH) OR VICE VERSA (REVERSE SWITCH).

Stable state:	Turbid water	buffering mechanisms	forward switches to 'clear water'
Factors contributing to stability	*Oscillatoria* bloom	resistant to low TP, low light and low temperature (1) reduced edibility for *Daphnia* grazers (2)	prolonged snow-covered ice (7) washout by winter flushing (8)
	phytoplankton bloom	bloom results in high pH, high sediment oxygen demand and thus high internal P loading, more blooms (3) bloom results in turbid waters, low piscivores, high planktivores, low grazing, more blooms (4)	control P release by sediment removal, sediment treatment or 'hard water' flushing (9) lower water level in spring to promote submerged vegetation (10) natural winter fish kills (11) reduce planktivores and promote piscivores (12)
	non-algal turbidity	wind-induced resuspension of sediments in plant-free lakes (5) fish-induced resuspension of sediments by benthivores, unhindered by plants (6)	reduce wind exposure of sediments (13) or complete drawdown and drying of sediments (14) reduce benthivores (15)
Stable state:	**Clear water**	**buffering mechanisms**	**reverse switches to 'turbid water'**
Factors contributing to stability	benthic diatoms	reduce susceptibility of lake sediments to wind-induced resuspension (16) compete with phytoplankton for N, P (17) promote N loss by denitrification (18)	benthivore stocking (26) storm events (27)
	submerged vegetation	competes with phytoplankton for N, P (19) promotes N loss by denitrification (20) reduces susceptibility of lake sediments to wind-induced resuspension (21) excretes substances allelopathic to phytoplankton (22) promotes grazing of phytoplankton by providing refuge to *Daphnia* (23) promotes phytoplankton grazing by providing refuge to pike and subsequent top-down control of planktivores (24) reduces fish-induced resuspension by hindering bottom feeding (25)	mechanical destruction of vegetation (28) chemicals toxic to vegetation (29) macrophyte grazing by birds (30) increase water level during spring (31) benthivore stocking (32) grass carp stocking (33) chemicals toxic to Daphnia (34) storm events (35)

TABLE 1 (continued). References: (1) (3) (8) (9): chapter 4 in Hosper, 1997; (2) (4) (6) (12) (15) (25): chapter 5 in Hosper, 1997; (5) (21): Jackson and Starret, 1959; James and Barko, 1990; Van den Berg *et al.*, 1996; (7): Greenbank, 1945; Raat, 1980; (10) (31): Holcomb *et al.*, 1975; Blindow *et al.*, 1993; Sanger, 1994; (11): Schindler and Comita, 1972; De Bernardi and Giussani, 1978; Raat, 1980; (13): Lürling *et al.*, 1995; (14): Cooke *et al.*, 1993; (16): Delgado *et al.*, 1991; (17) (18): Van Luijn, 1997; Van Luijn *et al.*, 1995; (19): Van Donk *et al.*, 1993); (20): Gumbricht, 1993; Van Donk *et al.*, 1993; (22): Wium-Andersen, 1987; Jasser, 1995; (23): Timms and Moss, 1984; Schriver *et al.*, 1995; Stansfield *et al.*, 1995; (24): Grimm, 1994; (26) (32): Ten Winkel, 1987; Crivelli, 1983; (27) (35): McKinnon and Mitchell, 1994; (28): - ; (29): Driessen *et al.*, 1993; (30): Van Donk, in press; (33): Small *et al.*, 1985; (34): Hurlbert, 1975; Shapiro, 1980; Scholten *et al.*, 1994.

How the concept of 'stable states, buffers and switches' can be used in lake restoration will be shown with the case study on the large (3,356 ha) and shallow (mean depth 1.25 m) Veluwemeer. Veluwemeer suffered from a massive bloom of cyanobacteria (*Oscillatoria agardhii*), persisting year-round. Early in 1979, the P loading of the lake has been reduced (from 2.7 to 1.5 g P m^{-2} y^{-1}) and from 1979-80 onwards, the lake has been flushed during winter with water low in TP and rich in Ca^{2+} and HCO_3^-. The winter flushing aimed at breaking the dominance of *Oscillatoria* and reducing the P release from the sediments. It was hypothesized that the *Oscillatoria* bloom supported a self-perpetuating process of algal activity, high pH, P release and even more algal activity (Hosper, 1997). As a result of the measures, TP and chlorophyll *a* showed a dramatic decline and algal growth during summer became P-limited (Fig. 5). Note the extremely low levels in 1996, probably due to the cold (long ice-cover) and dry (low external loading) preceding winter and spring (Figs. 12 and 13).

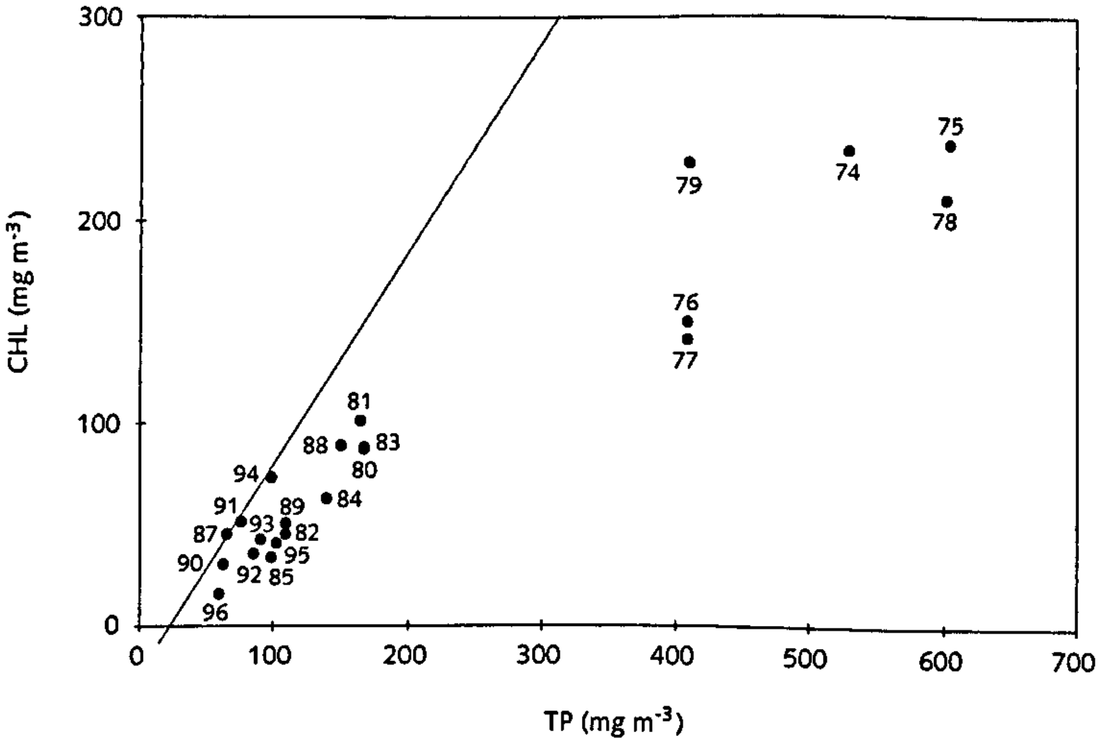


Fig. 5. Chlorophyll *a* in relation to TP for Veluwemeer, mean values April-September, 1974-1996.
The indicated line approximates the maximum value for chlorophyll *a* in relation to TP
(CHL = - 24 + 1.04 TP; Hosper, 1997).

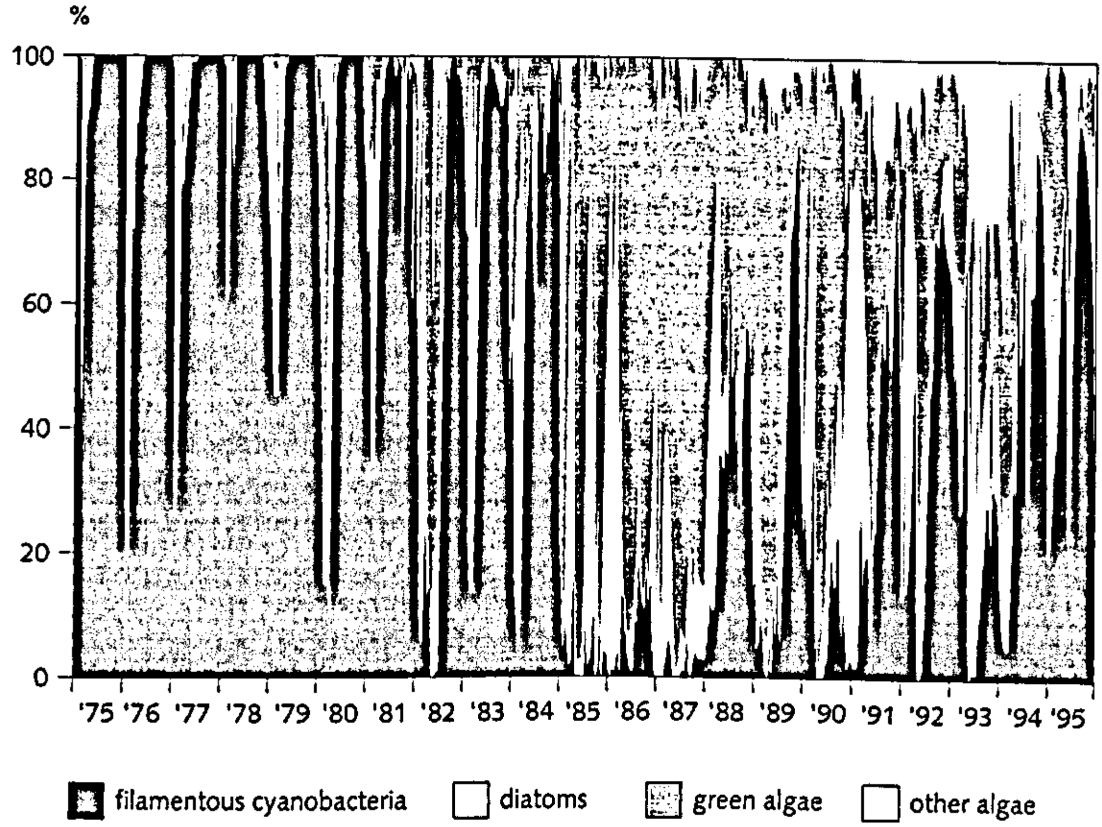


Fig. 6. Phytoplankton species composition in Veluwemeer, 1975-1995 (in % of total numbers).

For the first years, the intensity of winter flushing was insufficient to break the dominance of *Oscillatoria*. However, the cold winter of 1984-85 (Fig. 12) combined with the flushing, ultimately, resulted in a more diverse phytoplankton community (Fig. 6) with a varying portion of *Oscillatoria*. *Oscillatoria* disappeared from the phytoplankton (not shown) after the cold winter of 1995-96. Transparency during spring, in the beginning of the growing season, is particularly important to the growth of submerged macrophytes and therefore to the recovery process of the lake. A distinct 'spring clear water phase' usually occurs in lakes of moderate productivity, induced by zooplankton grazers (Sommer *et al.* 1986). Although transparency during spring in Veluwemeer showed an upward trend, Secchi depth in May-June was limited to 0.40-0.60 m only, with a peak of 0.80 m in 1996. Secchi depth during summer (April-September) increased from about 0.20 m to 0.30 m in the first years, and later after the species shift in phytoplankton to 0.40-0.50 m, increasing to 0.70 m in 1996 (Fig. 7).

From 1985 onwards, after the collapse of the *Oscillatoria* bloom, the sparse vegetation of *Potamogeton pectinatus* in the shallow parts (< 1 m) of the lake gradually became replaced by dense beds of Characeae (Coops *et al.*, 1996), covering about 25% of the lake in 1996 (Fig. 8). Surprisingly, the water overlying the *Chara* meadows is clear during summer, whereas the open water outside the vegetated area remains turbid (Fig. 9). Obviously, the two alternative stable states of macrophyte-dominated clear water and algae-dominated turbid water can coexist within one lake (Scheffer *et al.*, 1994; Van den Berg *et al.*, in press).
A similar development of clear vegetated areas and turbid open waters took place in the adjacent Wolderwijd, following fish stock reduction in 1991 (Meijer and Hosper, 1997). The final step in the restoration of Veluwemeer (and Wolderwijd) aims at increasing the clarity of the open water areas. The question now is whether this will happen spontaneously, resulting from a continuing expansion of the vegetation, or that additional measures (such as drastic fish stock reduction) are necessary for creating a more pronounced spring clear water phase. The favorable conditions in the summer of 1996 (Figs. 5, 7 and 8), following the extremely cold and dry winter (Figs. 12 and 13), suggest that the lake is close to a macrophyte-dominated

clear water state. After the growing season of 1996, transparency further increased and on 22 October the highest Secchi depth on record of 3.2 m was measured (Griffioen, pers. comm.). It makes sense to await further developments, before deciding upon additional actions, such as fish stock reduction. So far, it remains uncertain whether or not the present clear water lake will be resistant to occasional increases in external nutrient loading (e.g. resulting from a wet and mild winter). An increase in external loading may result in more algal biomass, which may trigger the positive feedback mechanism of P release from the sediments and even more algal biomass (Hosper, 1997). It can be hypothesized, however, that due to the relatively low external P loading since 1979, the binding of P to the sediments will have improved.

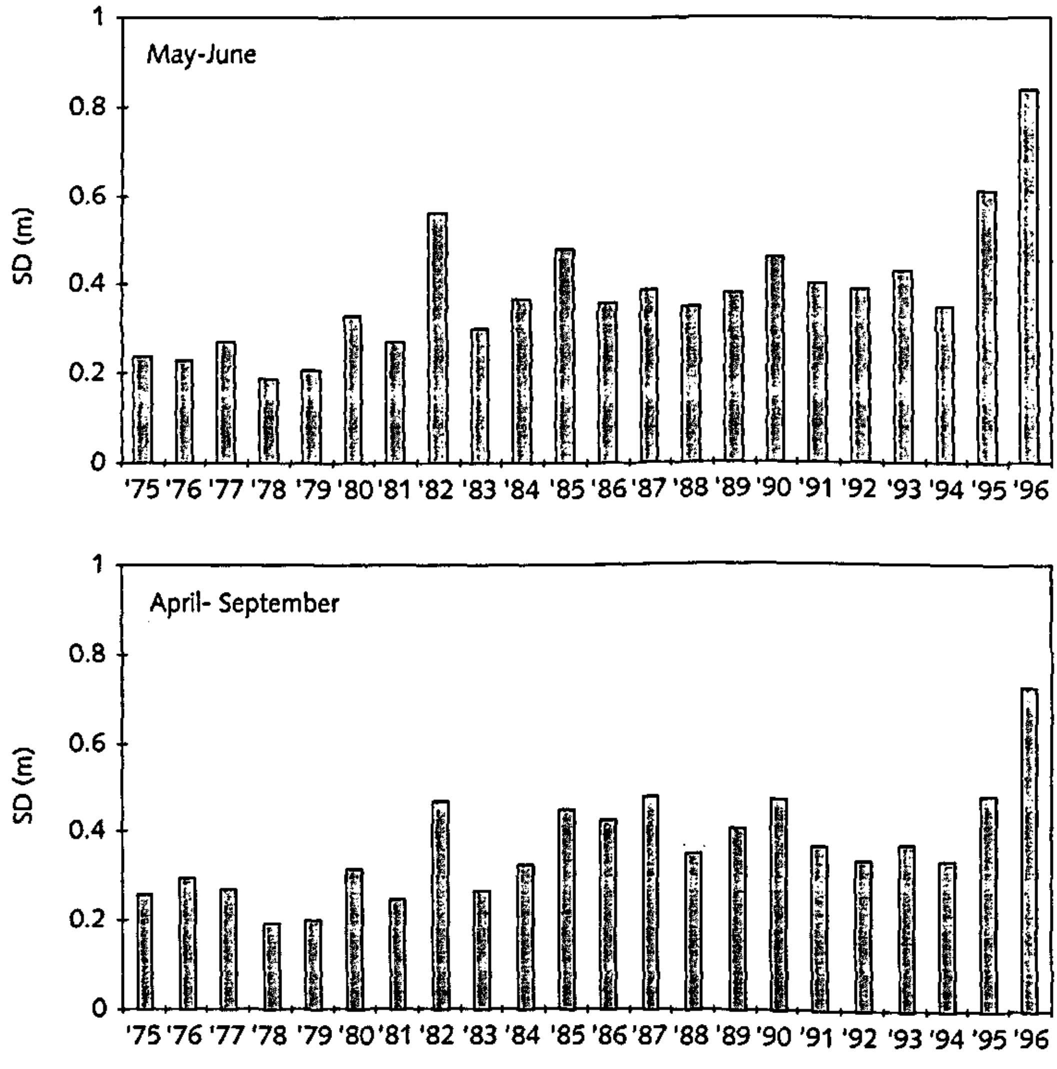


Fig. 7. Secchi depth in Veluwemeer (open water, unvegetated), mean values May-June (upper panel) and April-September (lower panel), 1975-1996.

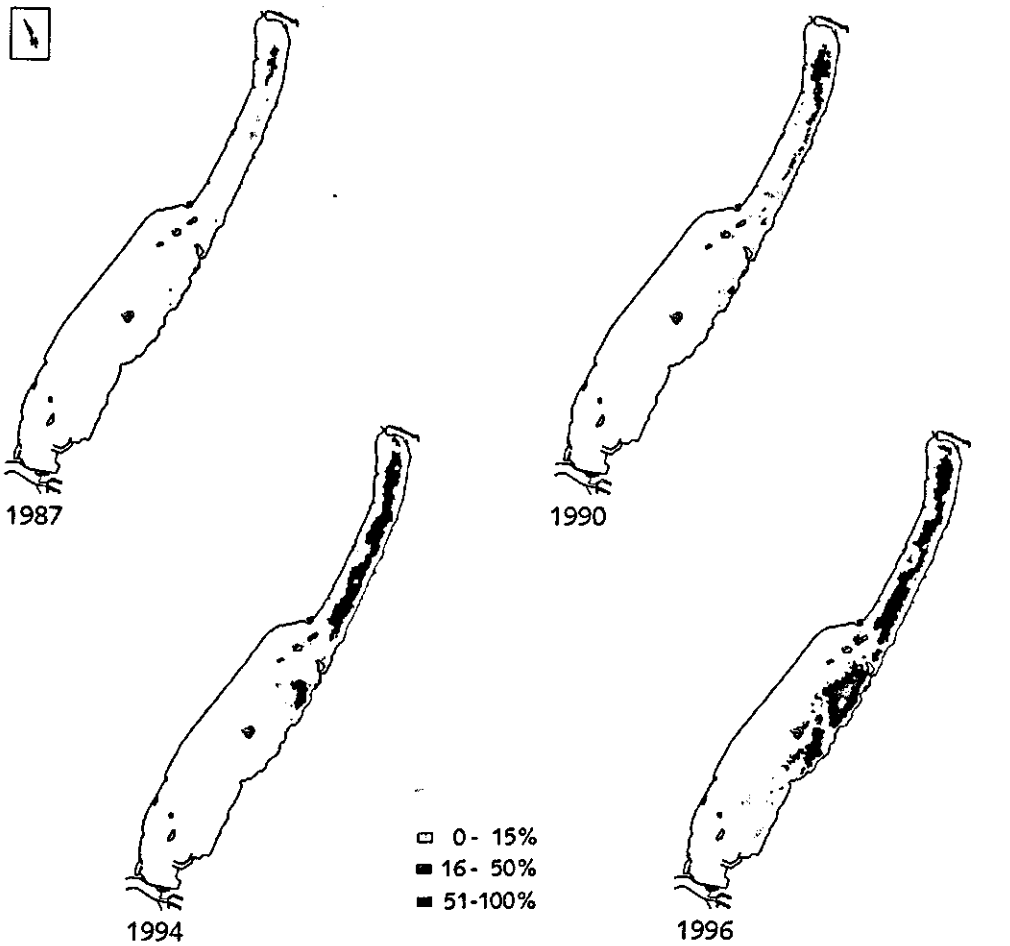


Fig. 8 . Characeae in Veluwemeer in 1987, 1990, 1994 and 1996. Density classes (% of bottom covered with vegetation) 0-15%, 16-50% and 51-100% (data from Doef *et al.*, 1991, 1994; De Witte *et al.*, 1995; RDIJ unpublished data).

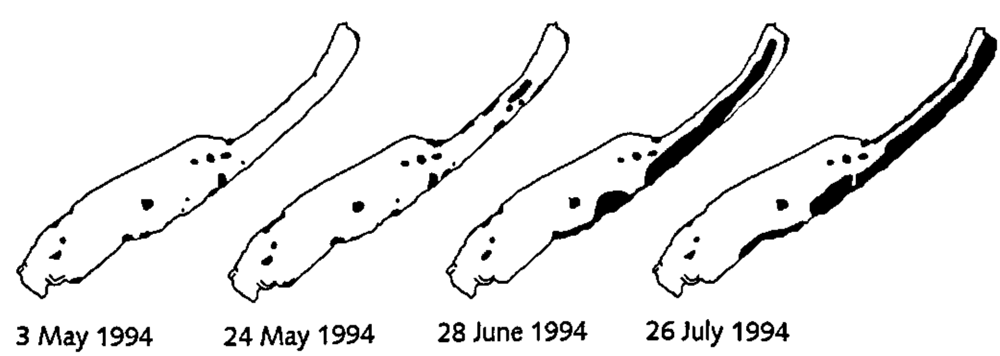


Fig. 9. Clear water areas in Veluwemeer in 1994. Shaded areas indicate bottom visibility from airplane observations (data from Zant *et al.*, 1995). May: unvegetated; June and July: densely vegetated.

In conclusion, several stages can be distinguished in the restoration process of Veluwemeer (Fig. 10). On top of the control of external loading, a series of 'switches' are hypothesized to lead to the ultimate goal of the macrophyte-dominated clear water state:

(1) 'Hard water' flushing
Control of internal P loading by manipulating the carbonate system, through winter flushing with water rich in Ca^{2+} and HCO_3^-. The winter flushing started in 1979-80 and as a result summer Secchi depth increased from 0.20 to 0.30 m.

(2) Washout of Oscillatoria by winter flushing during a cold winter
Breaking the dominance of the *Oscillatoria* bloom by winter flushing in combination with the cold winter 1984-85 (long periods with snow-covered ice). Summer Secchi depth in the open water increased from 0.30 to 0.40-0.50 m, and in the following years large shallow areas (20-30 % of the lake) became covered with a dense *Chara* vegetation with clear overlying water.

(3) Reduction of the fish stock
Promoting the macrophyte-dominated clear water state by creating a pronounced spring clear water phase, through drastic reduction of the planktivorous and benthivorous fish stock. Natural winter fish kills may also trigger the clear water state (Schindler and Comita, 1972; De Bernardi and Giussani, 1978; Haertel and Jongsma, 1982). However, for large lakes in the Netherlands such dramatic winter kills are very unlikely (Raat, 1980). As noted above, the favorable conditions in Veluwemeer (and in Wolderwijd) in the summer and fall of 1996, raise the question whether or not biomanipulation is still necessary.

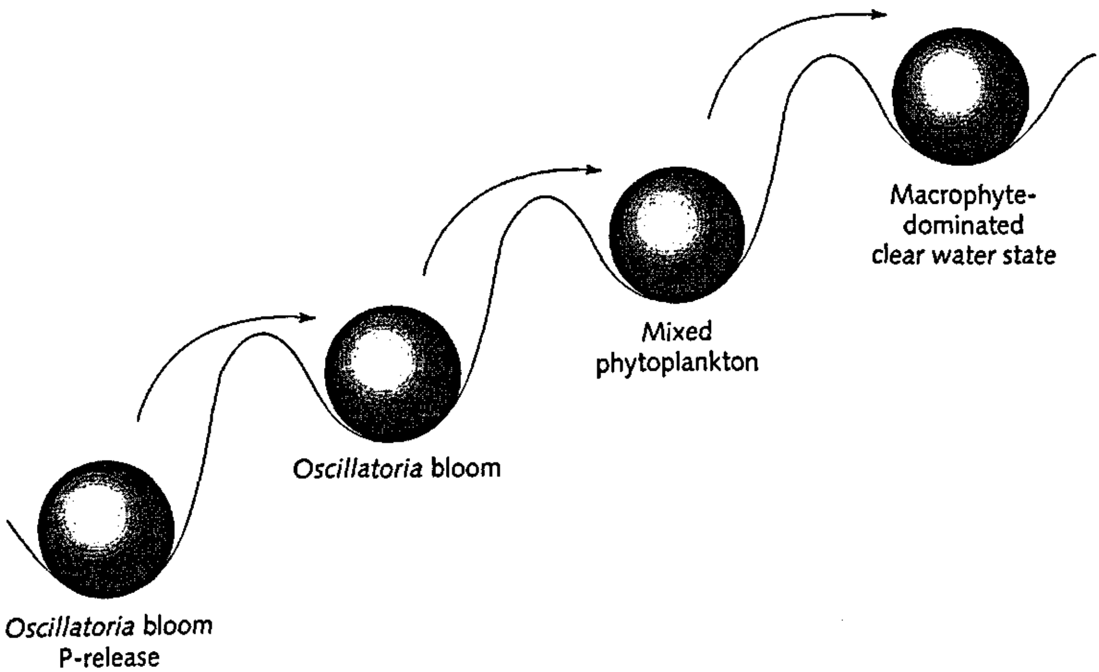


Fig. 10. The stepwise restoration process of Veluwemeer, in addition to the reduction of external P loading.
Step (1): 'hard water' flushing to control P release from the sediments; step (2): winter flushing to remove the *Oscillatoria* bloom and step (3): fish stock reduction to trigger a shift to the macrophyte-dominated clear water state.

Filamentous cyanobacteria (*Oscillatoria, Lyngbya*) blooming almost year-round, dominate the phytoplankton of numerous, shallow (z = 1-2 m) lakes in the Netherlands, making the water extremely turbid (Hosper, 1997). Deeper lakes (z = 2-4 m) are usually less turbid and show a mixed phytoplankton community with peaks of colonial scum-forming cyanobacteria (*Microcystis, Aphanizomenon*). Lake restoration primarily aims at clearing the water and controlling the scum-forming blooms of cyanobacteria (Anonymous, 1995; Van der Veer *et al.*, 1993, 1995). Here, attention is focused on the restoration of shallow lakes dominated by *Oscillatoria, Lyngbya* etc. (*Oscillatoria*-type algae). First of all, the primary cause of algal blooms, the increased external nutrient loading from point and non-point sources, should be eliminated. Furthermore, negative trends such as pollution with toxic chemicals (causing reduced zooplankton grazing) and the ongoing loss of lake-marginal wetlands and littoral vegetation (loss of pike habitat, loss of nutrient retention capacity) should be reversed.

Trends in TP, chlorophyll and phytoplankton species

TP levels in many surface waters in the Netherlands are going down, as a result of (inter-)national action programs. The TP flow in the Rhine river has been reduced by more than 50% (Anonymous, 1995). TP concentrations in the large (and relatively deep, z = 5 m) Rhine-fed lakes IJsselmeer and Markermeer (Fig. 11) have closely followed this trend. Obviously, for these lakes there is no delay in response due to the P release from the sediments. In IJsselmeer algal biomass tends to go down as well. In the adjacent Markermeer algal biomass usually is much lower and mainly light-limited, due to wind-induced sediment resuspension (Van Duin, 1992). Massive scum-forming blooms (e.g. *Microcystis aeruginosa*) still occur, particularly in IJsselmeer, during late summer (Ibelings, 1992).

The Friesland lakes, e.g. Tjeukemeer and Slotermeer (Fig. 11), showed strongly varying summer TP levels, which correlate with net rainfall during the first half of the year (Fig. 13). Low TP concentrations are observed in (extremely) dry years such as 1989, 1993 and 1996. The Friesland lakes receive water from the IJsselmeer (mainly in summer) and from the surrounding agricultural areas (mainly in winter). Apparently, P dynamics are largely controlled by on the one side P-rich drainage water pumped in from the agricultural areas during wet periods, and on the other hand the inlet of water from IJsselmeer, relatively low in TP, during dry periods. Summer chlorophyll *a* levels in the Friesland lakes were relatively low, particularly in Slotermeer, after the cold and dry winter and spring of 1996. *Oscillatoria* was virtually absent in the phytoplankton of Slotermeer, but was still the dominant species in Tjeukemeer, in the summer of 1996 (Maasdam, pers. comm.). Interesting developments showed up in De Wieden lake area, e.g. Beulakerwijde (Fig. 11). Chlorophyll *a* has strongly decreased, although TP levels remained relatively stable. Also in this lake area, TP and chlorophyll *a* were extremely low in summer 1996 and *Oscillatoria* practically disappeared after the cold winter (Moonen, pers. comm.). It has been suggested (Veldkamp, 1995; Hosper, 1997), that in De Wieden fish-eating cormorants could have played a significant role in the recent changes in the phytoplankton community. The Loosdrecht lakes showed a decline in TP and chlorophyll *a* (Fig. 11). However, the algal biomass is still relatively high, and the phytoplankton remains to be dominated by *Oscillatoria*-type algae (Everards, pers. comm.). The turbid water state of the Reeuwijk and Nieuwkoop lakes appears to be extremely stable. In both lakes the favorable 1996 weather conditions did not result in any response, neither in TP and chlorophyll *a* (Fig. 11), nor in the dominance of *Oscillatoria*-type algae (Frinking, pers. comm.). It was shown for Veluwemeer, in the previous section, that the cold winter of 1984-85 had contributed to the collapse of the *Oscillatoria* bloom in this lake. The earlier cold winter of 1978-79, however, failed in producing such a response in phytoplankton species (Fig. 6) or transparency (Fig. 7). Obviously, additional factors (e.g. lake flushing, reduced algal biomass) are necessary for the species shift. Winter 1995-96 triggered a major change in Veluwemeer, resulting in extremely clear water in the fall.

From the limited data set presented above, it is concluded that: (1) TP and chlorophyll *a* may vary strongly with weather conditions (Tjeukemeer, Slotermeer); (2) a cold and dry winter and spring, such as in 1996, may cause a collapse of the *Oscillatoria* bloom (Beulakerwijde), or even trigger a shift from turbid to clear (Veluwemeer); but (3) certain *Oscillatoria* lakes (Nieuwkoop, Reeuwijk, Loosdrecht) are more stable than others (Slotermeer, De Wieden).

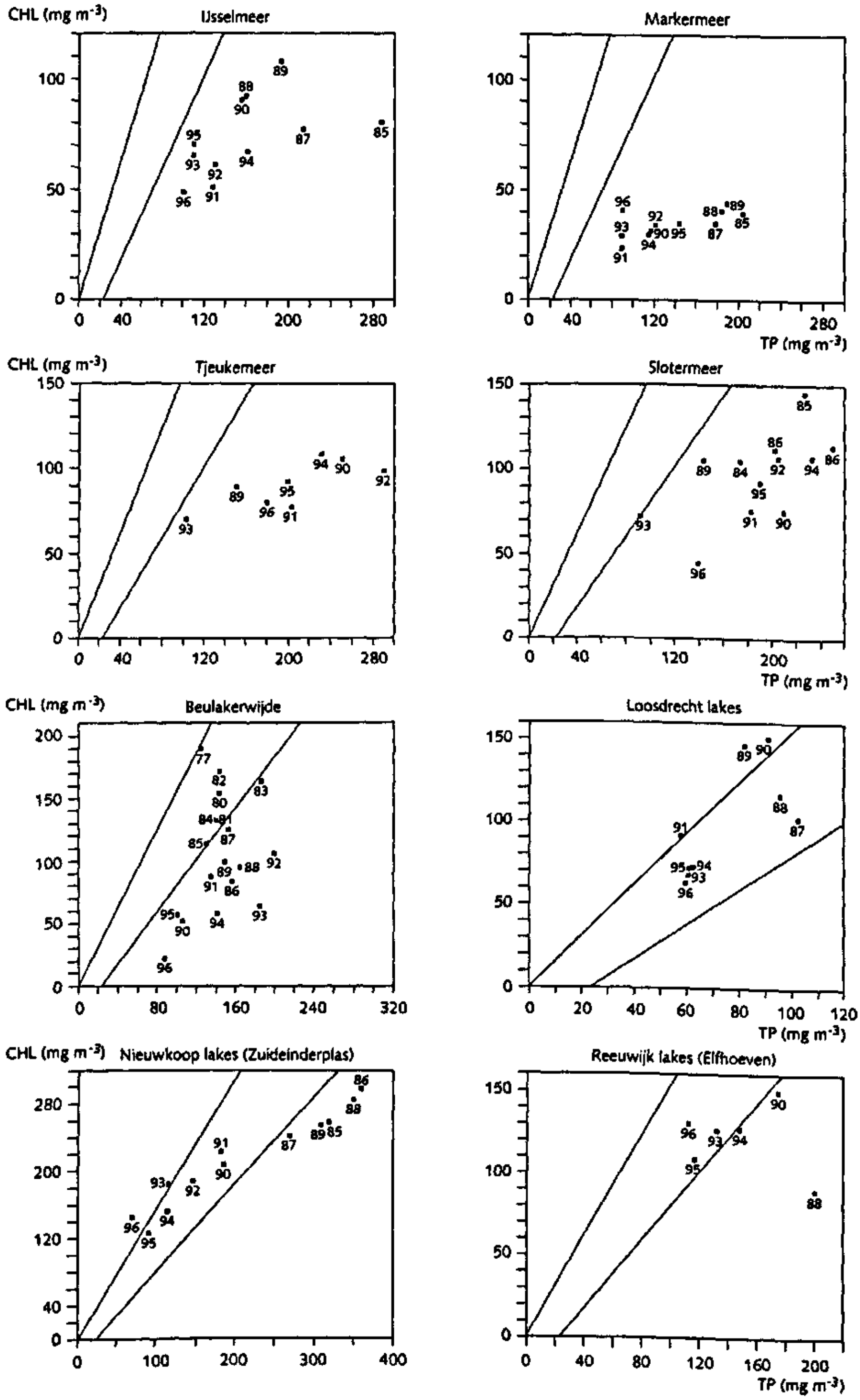


Fig. 11. Chlorophyll *a* in relation to TP for several lakes in the Netherlands, mean values April-September. The indicated lines approximate the maximum value for chlorophyll *a* in relation to TP for *Oscillatoria*-dominated lakes (CHL = 1.54 TP) and for other lakes (CHL = - 24 + 1.04 TP) (Hosper, 1997). Data from the lake management authorities.

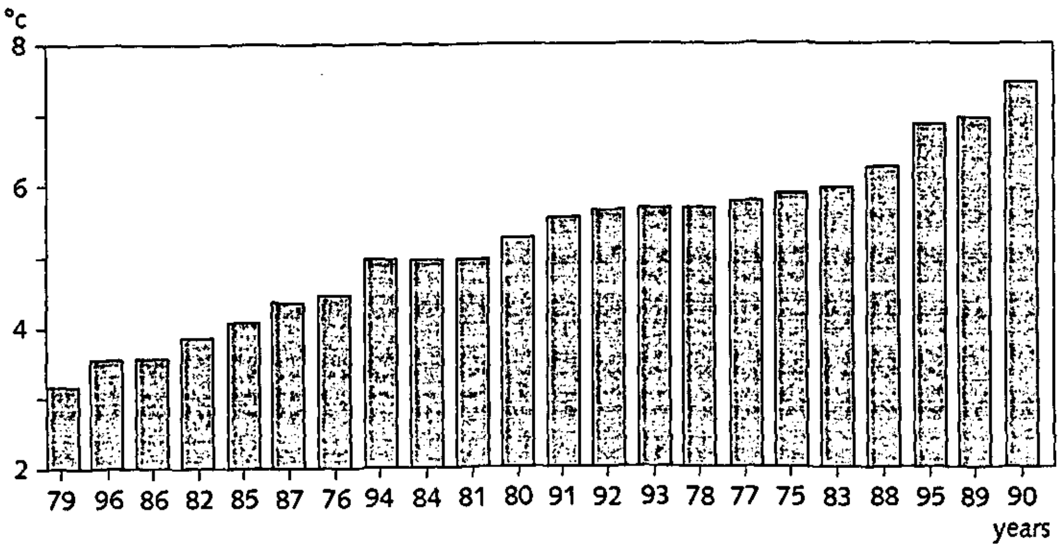


Fig. 12. Average air temperature in the Netherlands (at De Bilt) for October-March, 1975-1996.
Data from KNMI.

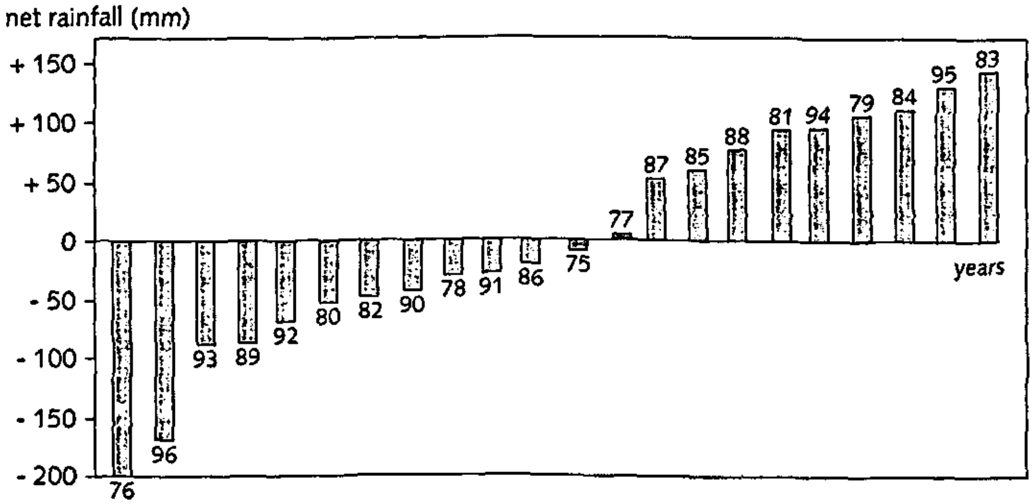


Fig. 13. Net rainfall (rainfall - evapotranspiration) in the Netherlands for January-June, 1975-1996.
Data from KNMI.

Additional in-lake measures

As the turbid water state tends to be resistant to lowering nutrient levels, additional in-lake measures may be necessary (TABLE 1). Certain blockages have to be removed to get the recovery process started: (1) the *Oscillatoria* bloom; (2) the algal bloom-mediated P release from the sediments and (3) the abundance of planktivorous and benthivorous fish, preventing *Daphnia* and submerged macrophytes from developing.

How can the Oscillatoria blooms be removed?
Oscillatoria blooms are particularly resistant to lake restoration efforts, due to the low edibility for *Daphnia* and the low P requirements. For many lakes it will be difficult to achieve TP < 20-50 mg m^{-3}, which is needed for a collapse of the bloom (Hosper, 1997). Therefore, additional switches are needed. A severe winter may contribute to the combat against *Oscillatoria*, as was shown in the previous section. Washout by winter flushing may be a powerful management tool, provided that flushing water is available in sufficient quantity and quality (low in TP, low in algae). It was concluded that conservative behavior (net growth = 0) may be assumed for *Oscillatoria* blooms, during November-February (Hosper, 1997). Thus, blooms can be effectively (> 95%) removed from well-mixed lakes by flushing in this four month period by at least three times the lake volume (flushing rate $\rho >$ 0.75 $month^{-1}$). The large lake IJsselmeer-Markermeer, which is relatively low in TP and algae (Fig. 11), may be a favorable source for flushing, for example the Friesland lakes or lakes in the western provinces. Small lakes in the Netherlands can possibly be flushed with clear water from adjacent deep sandpits (e.g. Reeuwijk lakes with water from the sandpit Broekvelden-Vettenbroek, or Holland-Ankeveen lakes with water from the sandpit Spiegelpolderplas).

How can P release from sediments be reduced?
Phytoplankton blooms may promote the P release from sediments by creating high sediment oxygen demand (SOD) and high pH. Such a self-perpetuating process (algal blooms, P release, more algal blooms) can be interrupted by flushing with water, which is low in TP and algae, but rich in Ca^{2+} and HCO_3^-. The manipulation of the carbonate system proved to be particularly successful in Veluwemeer (Hosper, 1997). Following 'hard water' winter flushing, the internal P loading strongly decreased, and algal growth was P-limited throughout the summer. For other lakes with high summer P release (apparent from high summer TP peaks), the significance and technical possibilities should be further evaluated. It has been suggested that the inlet of external water, which may be relatively rich in macro-ions such as bicarbonate, sulfate and chloride, into formerly isolated peat lakes, could enhance P release from the sediments (Smolders and Roelofs, 1995; Beltman and Van der Krift, 1997). For many peat lakes in the Netherlands, however, the naturally inflowing (ground)water already has relatively high concentrations of these macro-ions (see for example Van Liere *et al.*, 1989; Rip *et al.*, 1992; Hessels, 1995). Therefore, for these lakes the effects of the inlet of outside water on macro-chemistry are probably less dramatic than suggested, and the significance of altered macro-chemistry on P release may be questioned. Eiseltová (1994) concluded from work in the Swedish lake Trummen that dredging and removal of P-rich sediments from the lake system, though expensive, seemed to be the final answer to the problem of internal loading. However, sediment removal (25 -100 cm loose top layer) from the Dutch peat lake Geerplas, after severely reducing the external P loading, failed to result in the desired TP and phytoplankton levels (Van der Does *et al.*, 1992; Frinking and Van der Does, 1993). Initially, the sediment top layer in this lake was solid, but shortly afterwards a new loose mud layer had been formed. It has been demonstrated that P release after dredging was not significantly lower than before. Apparently, the new top layer shows the same P release characteristics as before the dredging (Van der Does and Frinking, 1993). Dredging is no panacea, and it seems wise to focus on external loading reduction and other more appropriate in-lake measures.

How can Daphnia grazing and macrophytes be restored?
Algae-dominated turbid lakes are usually rich in planktivorous and benthivorous fish, poor in piscivorous fish and practically devoid of submerged macrophytes. All these factors contribute to an unhindered development of algal blooms and stability of the turbid water state (TABLE 1). A single substantial reduction of planktivorous fish during winter may trigger a *Daphnia*-mediated clear water phase in spring, followed by a shift to a stable macrophyte-dominated clear water state (Hosper, 1997). Benthivore reduction, particularly in productive clay-bottom lakes, further supports the clearing of the lake. 'Inedible' filamentous cyanobacteria and the possible development of invertebrate predators (e.g. *Neomysis*), consuming the zooplankton grazers, are uncertain factors for spring clearing. Toxic compounds may also block the way to recovery and the lake water should be checked for acute or chronic toxicity to *Daphnia*. The formerly heavily polluted Rhine river has shown a dramatic recovery and currently toxic effects on *Daphnia*, reported for the 1970s (Sloof *et al.* 1985), can no longer be observed (Hendriks, 1995). Locally, however, particularly in intensively used agricultural areas, small water bodies (ditches) can still be acutely toxic (Gorter *et al.*, 1996). After fish stock reduction and clearing, rapid colonization of submerged macrophytes has been demonstrated in small lakes (< 30 ha)(Hosper, 1997). In large lakes recolonization takes more time and therefore repeated intensive fishing over several years may be necessary. In large lakes, and particularly in networks of interconnected lakes, it is more difficult to achieve a drastic reduction of the fish stock. Less drastic, but permanent winter fishing may be promising then, but case studies are needed for further evaluation. As noted above, prerequisites for successful biomanipulation are low TP levels (TP < 100 mg m^{-3}, or higher TP levels for very small lakes), and low numbers of inedible algal species, such as *Oscillatoria*. The chances for biomanipulation are improving, as in many large lake areas TP is approaching the indicated levels (Fig. 11). Further-

more, *Oscillatoria*-type algae are losing ground in large lake areas, including Veluwemeer (Fig. 6) and Wolderwijd (Meijer and Hosper, 1997), De Wieden (Moonen, pers. comm.) and some of the Friesland lakes (Maasdam, pers. comm.). Uncertainties remain, however, particularly for large shallow lakes. It should be noted that complete case studies covering the whole process of recovery from the stable turbid water state to the stable clear water state, are not available. Therefore, only tentative guidelines could be presented and several questions remain unanswered, such as: (1) the role of resuspended sediments in windswept lakes in preventing submerged vegetation from developing; (2) the role of submerged vegetation in maintaining clear water, not just in restricted areas, but throughout large shallow lakes and (3) the efficacy of repeated winter fishing in large and complex lake systems, for promoting *Daphnia*.

Priorities in lake restoration

Several authors have suggested radical measures for lake restoration, including the restoration of the original groundwater flow systems by reflooding adjacent polders (Engelen *et al.*, 1992; Verstraelen *et al.*, 1992; Barendrecht *et al.*, 1992), the construction of large-scale pike nursery habitats (Grimm, 1994; Klinge *et al.*, 1995), and restoring the former 'legakker' structure in peat lakes by constructing islands and dams (Lürling *et al.*, 1995; Everards, pers. comm.). Emulating the original natural processes, unquestionably, is the best guide to lake restoration. However, it will be clear that current economic and public interests may oppose to this approach and the lake manager has to look for more realistic tools (TABLE 2).

TABLE 2 PRIORITIES SUGGESTED FOR THE RESTORATION OF SOME SHALLOW DUTCH LAKES (* * * HIGH, * * MODERATE, * LOW PRIORITY).

	Reduction external P loading	Washout Oscillatoria by flushing	Promoting spring clear water phase by fishing	Additional measures
Veluwemeer	* *	*	* *	continue 'hard water' flushing
Wolderwijd	* *	*	* *	continue 'hard water' flushing
Friesland lakes	* * *	* *	* * *	-
De Wieden	* *	*	* *	protect cormorant colony
Loosdrecht lakes	*	* * *	* *	-
Nieuwkoop lakes	*	* * *	* *	-
Reeuwijk lakes	* *	* * *	* *	-

For many shallow lakes, including the Loosdrecht lakes, the Nieuwkoop lakes and the Reeuwijk lakes, it is obvious that the persistent bloom of *Oscillatoria*-type algae is the main bottleneck for recovery. Therefore, priority should be given to fighting the *Oscillatoria* blooms. Essentially, the options for eliminating *Oscillatoria* include (1) further reduction of external P loading or (2) improving the light conditions in the lake, so more favorable (edible) algal species could outcompete *Oscillatoria*, which prefers dim-light conditions. For the Loosdrecht lakes for example, it is feasible to bring the external loading further down from 0.35 to 0.10 g P m^{-2} y^{-1} (De Ruiter, 1992). Model calculations indicated that TP and chlorophyll *a* will decrease as a result, but that filamentous cyanobacteria will still be dominant (Janse *et al.*, 1992). The light conditions in the Loosdrecht lakes might be improved either by washout of the *Oscillatoria* (by flushing), or by reduction of non-algal turbidity. Wind-induced resuspension of sediments is the main cause of non-algal turbidity in the Loosdrecht lakes (Gons *et al.*, 1991) and would be extremely difficult to control. Radical (and costly) changes in the morphometry of the lake would be necessary (e.g. creating islands, dams or deep sediment accumulation basins). What remains is the option of flushing with water low in *Oscillatoria* and TP. Van Liere and Janse (1992) already evaluated the feasibility of flushing for the Loosdrecht lakes and concluded that the existing infrastructure does not allow an effective lake flushing. It is suggested here to explore new possibilities for flushing, including the use of mobile pumping engines to overcome infrastructural problems. Flushing possibilities should also be investigated for other persistent *Oscillatoria* lakes, such as the Nieuwkoop lakes and the Reeuwijk lakes.

The original natural water level fluctuations in the Friesland lakes usually resulted in the flooding of extended grassland areas during winter and spring (Klinge *et al.*, 1995). The flooded areas (temporal wetlands) acted as sinks for nutrients and fine sediment particles and as spawning and nursery habitat for pike, supporting the clear water state of the Friesland lakes at that time. Restoring such a flooding scenario will only be possible on a small scale (Claassen, pers. comm.). Grimm (1994) suggested creating of large-scale 'managed marshes' with an optimal water level regime for raising young pike. However, before deciding upon this kind of costly measures in the Friesland lakes, it seems wise to focus on the further control of the (non-point) nutrient loading and appropriate in-lake measures such as flushing and fishing. Winter flushing with water from IJsselmeer could be effective in fighting the *Oscillatoria* blooms. Winter fishing on bream and large pike-perch could enhance the top-down control of algae, through planktivorous fish and *Daphnia*. For the Friesland lakes this top-down control may be most promising, as the non-point P sources for these lakes (inflow of water from agricultural areas) will be difficult to control.

It was shown above that in De Wieden lake area (e.g. Beulakerwijde), *Oscillatoria* lost ground completely after the cold winter of 1996, and that cormorants possibly contributed to the top-down control of the algae. Winter fishing may further enhance this top-down control.

REFERENCES

Anonymous (1995). *Voortgangsrapportage Integraal Waterbeheer en Noordzee-aangelegenheden 1994.* Ministerie van Verkeer en Waterstaat (in Dutch).

Barendrecht, A., Stam, S.M.E., Wassen, M.J. (1992). Restoration of fen ecosystems in the Vecht river plain: cost-benefit analysis of hydrological alternatives. *Hydrobiologia 233*, 247-258.

Beltman, B., Van der Krift, T. (1997). De invloed van sulfaat en chloride op de fosfaatbeschikbaarheid in veenbodem, een bijdrage aan integraal waterbeheer. *H2O 30*, 19-22 (in Dutch).

Berger, C. (1987). *Habitat en ecologie van Oscillatoria agardhii Gomont.* PhD thesis, University of Groningen, the Netherlands (in Dutch).

Blindow, I., Andersson, G., Hargeby, A., Hansson, S. (1993). Long-term pattern of alternative stable states in two shallow eutrophic lakes. *Freshwater Biol.* 30, 159-167.

Coops, H. (1996). *Helophyte zonation: impact of water depth and wave exposure.* PhD thesis, Catholic University of Nijmegen, the Netherlands.

Coops, H., Doef, R.W., De Witte, B.J., Van den Berg, M.S. (1997). Herstel van de watervegetatie in de Veluwerandmeren. *De Levende Natuur 98*, 8-13 (in Dutch).

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

Crivelli, A.J. (1983). The destruction of aquatic vegetation by carp. *Hydrobiologia* 106, 37-41.

De Bernardi, R., Giussani, G. (1978). Effect of mass fish mortality on zooplankton structure and dynamics in a small Italian lake (Lago di Annone). *Verh. Internat. Verein. Limnol. 20*, 1045-1048.

De Deckere, E.M.G.T., Rijsdijk, R.E., Boers, P.C.M., Van der Molen, D.T. (1996). *Effecten van de reductie van fosfaatbelasting in ondiepe plassen en meren.* RIZA nota nr. 96.049, Lelystad. ISBN 9036950147 (in Dutch).

Delgado, M., De Jonge, V.N., Peletier, H. (1991). Experiments on resuspension of natural microphytobenthos populations. *Mar. Biol. 108*, 321-328.

De Ruiter, M.A. (1992). Water quality research in the Loosdrecht lakes: proposals for the follow-up restoration measures. *Hydrobiologia 233*, 161-163.

De Witte, B.J., Hector, L.H.C.A., Streekstra, M.L., Butijn, G.D. (1995). *Monitoring van waterplanten in het IJsselmeergebied in het kader van het regionaal meetnet (1990-1994).* RDIJ Intern rapport 1995-5 ANM/ANW (in Dutch).

Doef, R.W., Smits, A.J.M., Kerkum, F.C.M. (1991). *Water- en oeverplanten in het IJssselmeergebied (1987-1989).* RIZA nota nr. 90.015, Lelystad (in Dutch).

Doef, R.W., Coops, H., Streekstra, M.L., Hector, L.H.C.A. (1994). *Waterplanten in het Woldenvijd en het Veluwemeer (1990-1993).* RIZA nota nr. 94.046. ISBN 9036903440 (in Dutch).

Driessen, O., Pex, B., Tolkamp, H.H. (1993). Restoration of a lake: First results and problems. *Verh. Internat. Verein. Limnol. 25*, 617-620.

Eiseltová, M. (1994). *Restoration of Lake Ecosystems, a holistic approach.* A training handbook edited by Martina Eiseltová. IWRB Publication 32, Slimbridge, Gloucester, UK.

Engelen, G.B., Kal, B.F.M., Buyse, J.J., Van Pruissen, F.G.M. (1992). The hydrology of the Loosdrecht lakes area. *Hydrobiologia* 233, 21-38.

Frinking, L.J., Van der Does, J. (1993). Leidt baggeren tot een blijvende verbetering van de waterkwaliteit in de Geerplas? *H_2O 26*, 103-109 (in Dutch).
Golterman H.L. (ed) (1977). *Interactions between sediments and fresh water*. Proceedings of an international symposium held at Amsterdam, the Netherlands, September 6-10, 1976. Dr W. Junk B.V. Publishers the Hague, and Centre for Agricultural Publishing and Documentation, Wageningen. ISBN 90061935636, ISBN 9022006328.
Gons, H.J., Otten, J.H., Rijkeboer, M. (1991). The significance of wind resuspension for the predominance of filamentous cyanobacteria in a shallow, eutrophic lake. *Mem. Ist. ital. Idrobiol. 48*, 233-250.
Gorter, M., Van der Wal, B., Mangelaars, J. (1996). Veldtoets met watervlooien: een nuttige aanvulling op het chemisch onderzoek. *H_2O 29*, 354-357 (in Dutch).
Greenbank, J.T. (1945). Limnological conditions in ice covered lakes, especially as related to winter kill of fish. *Ecol. Monogr. 15*, 344-392.
Grimm, M.P. (1994). The characteristics of the optimum habitat of northern pike (*Esox lucius* L.). In: I.G. Cowx (ed), *Rehabilitation of freshwater fisheries*. Fishing News Books, Blackwell Scientific Publications Ltd. pp. 235-243.
Haertel, L., Jongsma, D. (1982). Effect of winterkill on the water quality of prairie lakes. *Proc. S. D. Acad. Sci. 61*, 134-151.
Hendriks, A.J. (1995). *Concentrations of microcontaminants and response of organisms in laboratory experiments and Rhine delta field surveys*. PhD thesis, University of Utrecht, the Netherlands.
Hessels, E. *Ontwikkeling van historische referentiebeelden voor enkele Nederlandse meren: Naardermeer, Loosdrechtse plassen, Reeuwijkse plassen en Oude Venen*. Stage opdracht LUW Wageningen/RIZA Lelystad (in Dutch).
Hofstra, J.J., Van Liere, L. (1992). The state of the environment of the Loosdrecht lakes. *Hydrobiologia 233*, 11-20.
Holcomb, D.E., Wegener, W.L., Williams, V.P. (1975). Lake level fluctuation for habitat management: a case in point. In: P.L. Brezonik and J.L. Fox (eds), *Proceedings of a symposium on water quality management through biological control*. University of Florida, Gainesville. pp. 151-157.
Hosper, S.H. (1997). *Clearing lakes, an ecosystem approach to the restoration and management of shallow lakes in the Netherlands*. PhD thesis, Agricultural University Wageningen, the Netherlands.
Hurlbert, S.H. (1975). Secondary effects of pesticides on aquatic systems. *Residue Rev. 58*, 81-148.
Ibelings, B.W. (1992). *Cyanobacterial waterblooms: the role of buoyancy in water columns of varying stability*. PhD thesis, University of Amsterdam, the Netherlands.
Jackson, H.O., Starrett, W.C. (1959). Turbidity and sedimentation at lake Chautauqua, Ill. *J. Wildl. Managem. 23*, 157-168.
James, W.F., Barko, J.W. (1990). Macrophyte influences on the zonation of sediment accretion and composition in a north-temperate reservoir. *Arch. Hydrobiol. 120*, 129-142.
Janse, J.H., Aldenberg, T., Kramer, P.R.G. (1992). A mathematical model of the phosphorus cycle in Lake Loosdrecht and simulation of additional measures. *Hydrobiologia 233*, 119-136.
Jeppesen, E., Jensen, J.P., Kristensen, P., Søndergaard, M., Mortensen, E., Sortkjær, O., Olrik, K. (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., Kristensen, P., Jensen, J.P., Søndergaard, M., Mortensen, E., Lauridsen, T. (1991). Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Mem. Ist. ital. Idrobiol. 48*, 127-148.
Klinge, M., Grimm, M.P., Hosper, S.H. (1995). Eutrophication and ecological rehabilitation of Dutch lakes: explanation and prediction by a new conceptual framework. *Wat. Sci. & Tech. 31 (8)*, 207-218.
Lürling, M., Blom, G.,Van Donk, E. (1995). Resuspensie, sedimentatie en lichtklimaat in de Breukeleveense Plas. *H_2O 28*, 194-198 (in Dutch).
McKinnon, S.L., Mitchell, S.F. (1994). Eutrophication and black swan (*Cygnus atratus* Latham) populations: tests of two simple relationships. *Hydrobiologia 279/280*, 163-170.
Meijer, M-L., Jeppesen, E., Van Donk, E., Moss, B., Scheffer, M., Lammens, E.H.R.R., Van Nes, E.H., Van Berkum, J.A., De Jong, G.J., Faafeng, B.A., Jensen, J.P. (1994). 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., Hosper, S.H. (1997). Effects of biomanipulation in the large and shallow lake Wolderwijd, the Netherlands. *Hydrobiologia 342/343*, 335-343.
Moss, B., (1983). The Norfolk Broadland: Experiments in the restoration of a complex wetland. *Biol. Rev. 58*, 521-561.
Moss, B. (1990). Engineering and biological approaches to the restoration from eutrophication of shallow lakes in which aquatic plant communities are important components. *Hydrobiologia 200/201*, 367-378.
Moss, B., Madgwick, J., Phillips, G. (1996). *A guide to the restoration of nutrient-enriched shallow lakes*. Environment Agency, Broads Authority. ISBN 0948119292.
Mur, L.R., Schreurs, H. (1995). Light as a selective factor in the distribution of phytoplankton species. *Wat. Sci. & Tech. 32*: 25-34.
Noordhuis, R. (ed) (1996). *Biologische monitoring zoete rijkswateren*. Watersysteemrapportage Randmeren. RIZA nota nr. 95.003 (in Dutch).
Raat, A. (1980). *Onderzoek wintersterfte 1978/79*. Literatuurapport. NVVS-OVB, Nieuwegein (in Dutch).
Rip, W.J., Everards, K., Houwers, A. (1992). Restoration of Botshol (The Netherlands) by reduction of external nutrient load: the effects on physico-chemical conditions, plankton and sessile diatoms. *Hydrob. Bull. 25*, 275-286.
Roelofs, J.G.M., Smolders, A. (1993). Major changes in aquatic peaty lowland ecosystems as a result of the inlet of River Rhine water. *De Levende Natuur 94*, 78-82 (in Dutch).
Sanger, A.C. (1994). The role of macrophytes in the decline and restoration of Lagoon of Islands. *Lake and Reservoir Management 9*, 111-112.
Scheffer, M., Hosper, S.H., Meijer, M-L., Moss, B., Jeppesen, E. (1993b). Alternative equilibria in shallow lakes. *Trend Ecol. Evolut.* 8, 275-279.

Scheffer, M., Van den Berg, M., Breukelaar, A., Breukers, C., Coops, H., Doef, R., Meijer, M-L. (1994). Vegetated areas with clear water in turbid shallow lakes. *Aquat. Bot. 49,* 193-196.

Schindler, D.W., Comita, G.W. (1972). The dependence of primary production upon physical and chemical factors in a small, senescing lake, including the effects of complete winter oxygen depletion. *Arch. Hydrobiol. 69,* 413-451.

Scholten, M.C.Th., Jak, R.G., Foekema, E.M. (1994). *Ecological control of algal densities. Towards a better comprehension of eutrophicational problems in aquatic ecosystems.* TNO, Delft, the Netherlands, TNO-MW-R94/280.

Schreurs H. (1992). *Cyanobacterial dominance, relations to eutrophication and lake morphology.* PhD thesis, University of Amsterdam, the Netherlands.

Schriver, P., Bøgestrand, J., Jeppesen, E., Søndergaard, M. (1995). Impact of submerged macrophytes on fish - zooplankton - phytoplankton interactions: large-scale enclosure experiments in a shallow eutrophic lake. *Freshwater Biol. 33,* 255-270.

Shapiro, J., Lamarra, V., Lynch, M. (1975). Biomanipulation: an ecosystem approach to lake restoration. In: P.L. Brezonik and J.L. Fox (eds), *Proceedings of a symposium on water quality management through biological control.* University of Florida, Gainesville. pp. 85-96.

Shapiro, J. (1980). The importance of trophic level interactions to the abundance and species composition of algae in lakes. In: J. Barica and L.R. Mur (eds), *Hypertrophic ecosystems.* Developments in Hydrobiology, Vol. 2, Junk, The Hague. pp. 105-116.

Sloof, W., Canton, J.H., Van der Gaag, M.A. (1985). Betekenis van tien jaar ecotoxicologisch onderzoek aan Rijnwater. *H_2O 18,* 119-126 (in Dutch).

Small, J.W., Jr., Richard, D.I., Osborne, J.A. (1985). The effects of vegetation removal by grass carp (*Ctenopharyngodon idella)* and herbicides on the water chemistry of four Florida lakes. *Freshwater Biol. 15,* 587-596.

Smolders, A.J.P., Roelofs, J.G.M. (1995). Internal eutrophication, iron limitation and sulphide accumulation due to the inlet river Rhine water in peaty shallow waters in the Netherlands. *Arch. Hydrobiol. 133,* 349-365.

Sommer, U., Gliwicz, Z.M., Lampert, W., Duncan, A. (1986). The PEG-model of seasonal succession of planktonic events in freshwaters. *Arch. Hydrobiol. 106,* 433-471.

Stansfield, J.H., Perrow, M.R., Tench, L.D., Jowitt, A.J.D., Taylor, A.A.L. (1995). Do macrophytes act as refuges for grazing Cladocera against fish predation. *Wat. Sci. & Tech. 32,* 217-220.

Ten Winkel, E.H. (1987). *Chironomid larvae and their foodweb relations in the littoral zone of Lake Maarsseveen.* PhD thesis, University of Amsterdam, the Netherlands.

Timms, R.M., Moss, B. (1984). Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. *Limnol. Oceanogr. 29,* 472-486.

Van den Berg, M.S., Coops, H., Meijer, M-L., Scheffer, M., Simons, J. (in press). Clear water associated with a dense *Chara* vegetation in the shallow and turbid Lake Veluwemeer, the Netherlands. In: E. Jeppesen, *et al.* (eds), *The role of submerged macrophytes in structuring the biological community and bio-geochemical dynamics in lakes.* Workshop 16-20 June, 1996. Silkeborg, Denmark.

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

Van der Does, J., Frinking, L.J. (1993). Effects of dredging of the phosphorus enriched upper sediment layers and reduction of the external loading in lake Geerplas. In: G. Giussani and C. Callieri (eds), *Proceedings Strategies for lake ecosystems beyond 2000.* 5th International Conference on the Conservation and Management of Lakes. 17-21 May, 1993. Stresa, Italy. pp. 199-202.

Van der Veer, B., Van Nieuwenhuyze, R.F., Donze, M. (1993). Accumulation of blue-green algal scums in small harbours and its prevention. *Verh. Internat. Verein. Limnol. 25:* 610-613.

Van der Veer, B., Koedood, J., Visser, P.M. (1995). Artificial mixing: a therapy measure combating Cyanobacteria in Lake Nieuwe Meer. *Wat. Sci. & Tech. 31 (8),* 245-248.

Van Donk, E. (in press). Switches between clear water state and turbid state induced by herbivory on macrophytes. In: E. Jeppesen, *et al.* (eds), *The role of submerged macrophytes in structuring the biological community and bio-geochemical dynamics in lakes.* Workshop 16-20 June, 1996. Silkeborg, Denmark.

Van Donk, E., Gulati, R.D., Iedema, A., Meulemans, J.T. (1993). Macrophyte-related shifts in the nitrogen and phosphorus contents of the different trophic levels in a biomanipulated shallow lake. *Hydrobiologia 251,* 19-26.

Van Donk, E., Gulati, R.D. (1995). Transition of a lake to turbid state six years after biomanipulation: mechanisms and pathways. *Wat. Sci. & Tech. 32,* 197-206.

Van Duin, E.H.S. (1992). *Sediment transport, light and algal growth in the Markermeer.* PhD thesis, Agricultural University Wageningen, the Netherlands.

Van Liere, L., Breebaart, L., Kats, W., Buyse, J.J. (1989). De waterkwaliteit in het Loosdrechtse plassengebied. In: *Integraal waterbeheer in het Goois/Utrechts stuwwallen- en plassengebied.* Redactie L. van Liere, R.M.M. Roijackers en P.J.T. Verstraelen. CHO-TNO nr. 22, Den Haag. pp. 265-279 (in Dutch).

Van Liere, L., Gulati, R.D. (1992). Restoration and recovery of shallow eutrophic lake ecosystems in The Netherlands: epilogue. Proceedings of a conference held in Amsterdam, The Netherlands, 16-19 April 1991. *Hydrobiologia 233,* 283-287.

Van Liere, L., Janse, J.H. (1992). Restoration and resilience to recovery of the Lake Loosdrecht ecosystem in relation to its phosphorus flow. *Hydrobiologia 233,* 95-104.

Van Luijn, F. (1997). *Nitrogen removal by denitrification in the sediments of a shallow lake.* PhD thesis, Agricultural University Wageningen, the Netherlands.

Van Luijn, F., Van der Molen, D.T., Luttmer, W.J., Boers, P.C.M. (1995). Influence of benthic diatoms on the nutrient release from sediments of shallow lakes recovering from eutrophication. *Wat. Sci. & Tech.* 32: 89-98.

Van Vierssen, W., Hootsmans, M.J.M., Vermaat, J.E. (eds) (1994). *Lake Veluwe, a macrophyte-dominated system under eutrophication stress.* Geobotany 21. Kluwer Academic Publishers. ISBN 0792323203.

Veldkamp, R. (1995). Diet of cormorants *Phalacrocorax carbo sinensis* at Wanneperveen, the Netherlands, with special reference to bream *Abramis brama. Ardea 83,* 143-155.

Verstraelen, P.J.T., Wisserhof, J., Rodic Lj., Eijsink, R. (1992). Eutrophication control strategies for three shallow Vecht lakes in the

province of North Holland. *Hydrobiologia 233*, 235-246.
Vollenweider, R.A. (1968). *Scientific fundamentals of the eutrophication of lakes and flowing waters, with special reference to nitrogen and phosphorus as factors in eutrophication.* Tech. Rep. OECD Paris. DAS/CSI/68, 27, 1-182.

USE OF MODELS IN EUTROPHICATION MANAGEMENT: THE KIS-BALATON CASE

L. Somlyódy
Budapest University of Technology, Müegyetem rkp.3, 1111 Budapest, Hungary

ABSTRACT

Models are often used for eutrophication management of lakes, reservoirs and other surface waters. Main shortcomings are related (i) to the uncertain cause-effect relations of mostly shallow, hypertrophic, blue-green algae dominated systems of large internal load which make future predictions difficult and (ii) complex decision making process of conflicting interests, of which eutrophication control may be only one element. The Kis-Balaton Reservoir (in Hungary), the subject of the present paper, is characterized by both features. The reservoir system consisting of two elements was designed to reduce the phosphorus load of agricultural non-point source origin to control the eutrophication of Lake Balaton. The original assumption of the late seventies was that macrophytes will play the decisive role in nutrient removal. Operational experiences and the ongoing unique monitoring program justified that this was not the case: the Upper Reservoir quickly developed into a hypertrophic phytoplankton system. The P retention of the inundated reed area of the Lower Reservoir is not satisfactory and the large, intact reed stand of high nature conservation value started to die off due to the water level increase and algal-rich inflow. Thus, the need for the revision of the original design (more than half of which was implemented) became evident. The paper summarizes the approach adopted, project alternatives and the criteria of evaluation, details of simple and complex P cycle models applied for the Upper Reservoir and their transfer to the lower one to derive estimates for the future P retention. It also discusses dilemmas related to the estimation of the sorption isotherm of the soil/sediment system crucial to "extrapolate the future" and outlines recommendations prepared for modifying the original plan.

KEY WORDS

Balaton, eutrophication, inundation, Kis-Balaton, management, modeling, operation, policy making, sediment, soil, sorption, water quality models.

INTRODUCTION

In the course of past decades models have been increasingly used to understand and manage eutrophication of surface waters. Lake eutrophication models range from simple steady state, Vollenweider type of empirical total phosphorus models to complex simulation ones (see e.g. OECD, 1982, Somlyódy and van Straten, 1986, Thomann and Mueller, 1987, Straskraba, 1995). Another cluster of models deals with watershed nutrient loads, primarily with diffuse sources (see e.g. Olem and Novotny, 1994), where again a number of different tools exist. If a chosen nutrient load model is coupled to a lake eutrophication model of a certain kind and costs, benefits and other evaluation criteria are also considered, a single or multiple criteria management optimization model is resulted. Examples exist, although not too many (see e.g. Somlyódy and Wets, 1988, Somlyódy, 1997).

Any of the above tools - including the traditional Vollenweider relationship - which address the issue of reducing loads to meet certain future trophic state goals can be considered as eutrophication management models. A particular load reduction can be realized in many different ways, depending where and when measures are taken in the watershed, i.e. what is the strategy. This is where conflicting, often hidden interests and non-quantifiable factors may enter the picture, eventually in the frame of a decision or policy making process. "Technical" analyses performed by using models is just one element of the procedure. It is often said that modeling is an "art". Decision making is another, different "art", and thus a model based decision support is a "double art".

Advanced computer technology enables us to put together relatively easily complex decision support systems for eutrophication management. However, the "bottleneck" of management and policy making is different. *One of the serious difficulties* stems from our limited understanding on the behavior of existing systems which serve as a basis to design load reduction programs (i.e. to predict the "future"): cause-effect relations are often poor or missing. For instance, this is the case for Lake Balaton (Hungary) for which the internal phosphorus load and the dominance of the nitrogen fixing blue-green algae have been causing positive and negative "surprises" in the course of the past two decades. Another example is Lake Paranoa, a man made hypertrophic lake in Brasilia where P retention could be best characterized by an "immediate loss" of P downstream to discharges of shallow riverine side arms (probably due to adsorption on the soil/sediment of high iron content, Somlyódy et al., 1989). How this loss changes due to wastewater treatment, or in other cases how the apparent settling rate of "popular" simple models (or calibrated parameters of a complex model) depends on load reductions, is often unknown. Thus, frequently we can not do more than "intelligent speculations". This is particularly true if we handle shallow, hypertrophic lakes characterized by the presence of blue-green algae and large internal P load.

The second difficulty comes from the "double art" nature of policy making already referred to: the policy implemented may not agree with the suggested, "feasible" one, in which respect the poor performance of models and the mistrust stemming from it also play a role. In addition, eutrophication management is often just one element of a complex water resources issue of many diverse criteria and conflicts where the development of an acceptable compromise solution may not be easy.

The objective of the present paper is to demonstrate some of the above shortcomings on the example of a shallow, hypertrophic system, the Kis-Balaton Reservoir in Hungary. We will outline how nutrient models were used to "predict" P retention, what were the main dilemmas, criteria of the evaluation of alternatives and some of the recommendations developed for policy making. The case also illustrates the importance of project preferences changing with time.

THE KIS-BALATON RESERVOIR SYSTEM: PROBLEM FORMULATION AND APPROACH

Background

The establishment of the Kis-Balaton Reservoir as a pre-impoundment system (Fig. 1) was an element of the regional P load control strategy worked out for Lake Balaton (about 596 km^2 surface area and 3.1 m average depth) in the early 1980s and approved by the government in 1983 (Somlyódy and van Straten, 1986).

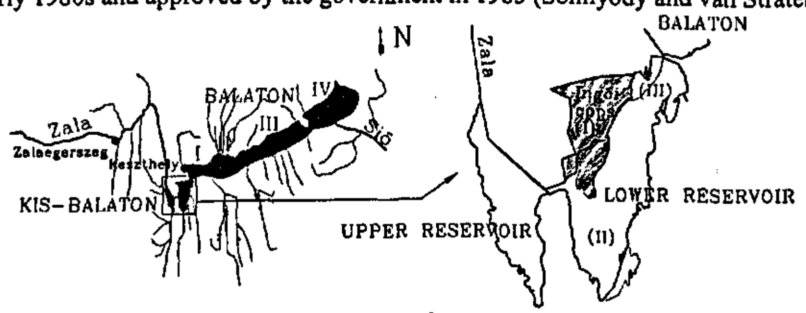


Fig. 1 Lake Balaton and the Kis-Balaton Reservoir

The major aim of the Kis-Balaton was to reduce primarily agricultural non-point source pollution from the Zala catchment forming about half of the lakes total watershed and loading the most western, smallest

Keszthely-basin. Basin I. (Fig. 1) became hypertrophic already at the late seventies which was followed by the other ones with a time lag, though their state remained significantly better.
The idea of "reconstructing" the Kis (Small)-Balaton (which formed the most western basin of the lake prior to lowering the water level by about 2 m in the last century) consisting of the upper and lower elements (1.1 m average depth, 18 km^2 and 50 km^2 surface area, and 20 days and 50 days filling time) emerged nearly three decades ago. The original concept assumed that the reservoir will act primarily as a wetland "filter" which removes a high portion of nutrients - eventually by harvesting macrophytes (see Kárpáti et al., 1980).

The problem

The engineering design of the reservoir was prepared in the late seventies. The Upper Reservoir (Fig.1) was inundated at once in 1985. In the Lower Reservoir partial flooding followed by a gradual water level increase was launched later than planned, only in 1992. More importantly, P precipitation at the municipal wastewater treatment plant of Zalaegerszeg (Fig. 1) - the largest town in the region - started to operate much later than scheduled by the 1983 policy, only in 1991 which led to a significant overload of the Upper Reservoir.

In order to follow water quality changes of the Zala River and future impacts of the planned reservoir, a unique observation program was initiated in 1976 which included daily measurements at several locations. According to the data, the Upper Reservoir has quickly developed into a hypertrophic phytoplankton dominated lake (i.e. in contrast to the original concept, macrophytes have disappeared) that has retained its phosphorus load with a reasonable efficiency until 1991. After introducing P precipitation at Zalaegerszeg, P retention of the Upper Reservoir has significantly decreased (see later).

Measurements of the partially flooded Lower Reservoir, the so-called Ingói copse with an area of 16 km^2 (Fig. 1) reveal that retention of the biologically available P is not satisfactory. Moreover, the area functions as a net source of phosphate. A literature survey shows that our understanding of nutrient cycling in macrophyte dominated systems - if similar systems exist at all - is rather weak, the functioning of wetlands, including their P retention is hardly predictable and except adsorption on soil/sediment, perhaps no other P removal processes can be identified.

Nowadays the importance of nature protection is increasingly recognized in general, as well as specifically in the Kis-Balaton region. The latter is a nature conservation area and the reservoirs themselves are strictly protected areas under the auspices of the Ramsar Convention. The most valuable is the Ingói copse of a large, intact reed stand, exactly the area which is characterized by an uncertain nutrient retention (the section close to the mouth is also dominated by a reed stand which, however, is aged and less valuable). Maintenance of this stand that represents a unique value as well as prevention of strong water level fluctuations during the nesting period of the waterfowl have become essential requirements. The increase of water depth and algal-rich inflow water already started to induce die-back symptoms of the reed stands.

Thus, there was an obvious need for the revision of the existing plan. Herodek and Somlyódy (1996) submitted a proposal on the necessity and methodology of revision. They defined the objective of the revision as selection of solution(s) that would make a compromise between the interests of nutrient retention, nature conservation and flood protection, and that would take into consideration the present level of project realization (more than 50 %) as well as expected additional costs. The revision - following the decisions of the competent governmental institutions - began in the autumn of 1996 and was completed early 1997.

Approach, alternatives and criteria

The approach developed for the broad decision making problem incorporated the integrated study of nutrient cycling in the Upper and Lower Reservoirs (Fig. 1), soil and sediment processes, groundwater, water balance, water circulation by a 2D model, historical water level fluctuations and so on (for details see Somlyódy et al., 1997). Large number of alternatives were developed and they were evaluated under nutrient load and hydrologic scenarios. Criteria of the evaluation included (i) TP retention, (ii) nature conservation, (iii) flood protection, (iv) incremental investment and OMR costs, (v) level of uncertainties and (vi) the sophistication of the operation (for criteria (i), (iii) and (iv) natural units, while for (ii), (v) and (vi) a five class ranking was employed).

Main alternatives depended on (i) planned functions of the limnologically and hydrobotanically rather different three regions (which can be nutrient removal, flood protection and nature conservation, Fig. 1); (ii) the extension of the above areas; (iii) the final water level and the rate of inundation to reach it; and (iv) accordingly, whether a macrophytes dominated wetland or an open water was envisaged. Further alternatives were obtained depending on (v) the diversion of a major stream from north to Area (I) or (II) which carries relatively high loads including the treated wastewater of the city Keszthely (Fig. 1) and the level of upgrading of P removal (the present TP effluent concentration is 1.8 mg/l), and (vi) the way of implementing a by-pass channel for the Zala River. This was suggested to protect Lake Balaton from the likely internal P load of the Lower Reservoir during the first few summers (see later) and to ensure overall flexibility of the operation.

The review of the wetland and limnology literature, as well as the detailed study of processes of the Kis-Balaton led to the conclusion (see later) that (i) at least a portion of the Lower Reservoir should be intentionally shifted towards algae eutrophication, the P removal of which is relatively well understood (in comparison to wetlands) and (ii) in this respect the Upper Reservoir serves as the closest case study. Thus, we calibrated P cycle models for the Upper Reservoir by using the exceptionally fine resolution data and transferred them to the Lower Reservoir. It was felt that the introduction of P removal in Zalaegerszeg (as well as the impact of droughts and the reduction in fertilizer application) caused such a significant load disturbance which should guarantee the identification of the sediment compartment crucially needed to estimate longer term behavior of the reservoirs (see later).

ANALYSIS OF PHOSPHORUS RETENTION

Upper Reservoir

As a first step, the shallow lake version of the Vollenweider model was applied. Fig. 2 illustrates TP removal efficiencies (see also Fig. 3 and Clement et al., submitted for details). The poor performance is apparent, and in fact mass balance calculations justify the significant decrease of the apparent settling rate from 1990 onwards the reason of which may be the appearance of internal load.

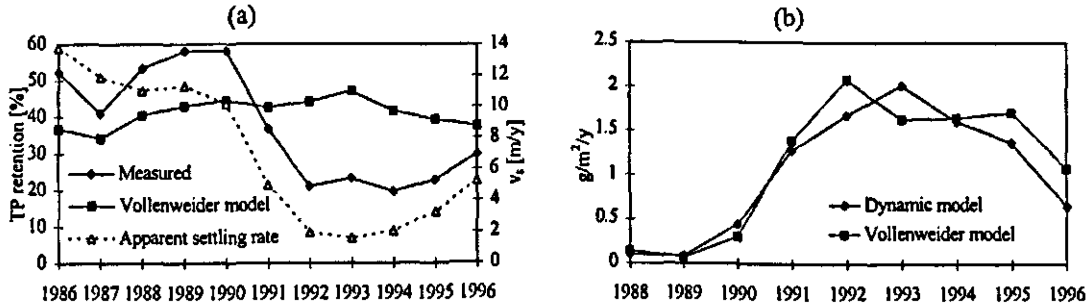


Fig. 2 Upper Reservoir: (a) TP removal efficiency and apparent settling rate; and (b) Estimated internal P load

A simple interpretation of the observations shows a discontinuous change of the output TP load (Fig. 3 (a)): all the points approximately follow the same linear behavior, but the $TP_{out} = f(TP_{in})$ relation exhibits a shift by about 30 t/y after 1990/1991, which is roughly equivalent to the reduction of the external load.

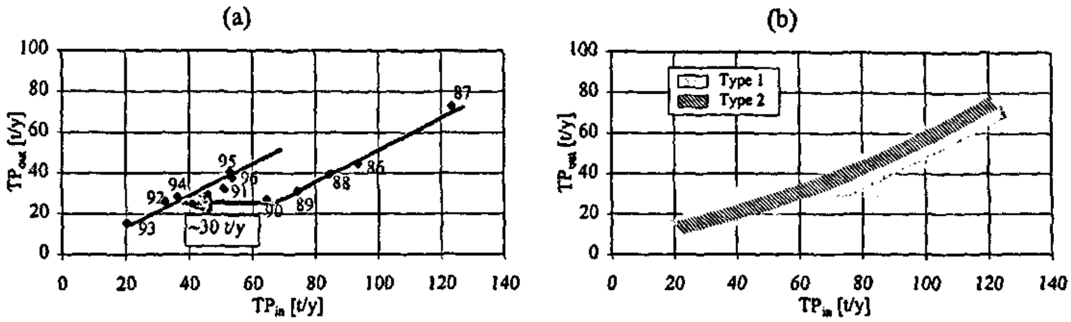


Fig. 3 Outflow load vs the inflow one (a) and types of model behaviors (b)

The basis of explanation lies in Fig. 4. In spite of the decrease in the external load, primary production and algae biomass remained practically unchanged (1200-1500 gC/m^2/y, a very high) which is possible only if the net "adsorption" before 1991 is replaced by a net "desorption". In other words, the external P load reduction leads on the short run to an internal load (which is in harmony with experiences with other lakes, see Sas, 1990) such that the total load remains practically unchanged. This load is determined by P needs of the phytoplankton and abiotic processes affecting the "equilibrium" of dissolved reactive P (DRP). The DRP concentrations at the outflow are controlled in a narrow range (below 30 mg/m^3) and the phytoplankton is P-sufficient. *These jointly lead to the hypothesis that biological P uptake continuously forces the DRP* concentration below the "sorption" equilibrium towards the threshold P concentration of net uptake by the organisms which is then compensated for by P desorption from the resuspended sediments (Lijklema et al., 1986, Istvánovics and Somlyódy, 1997).

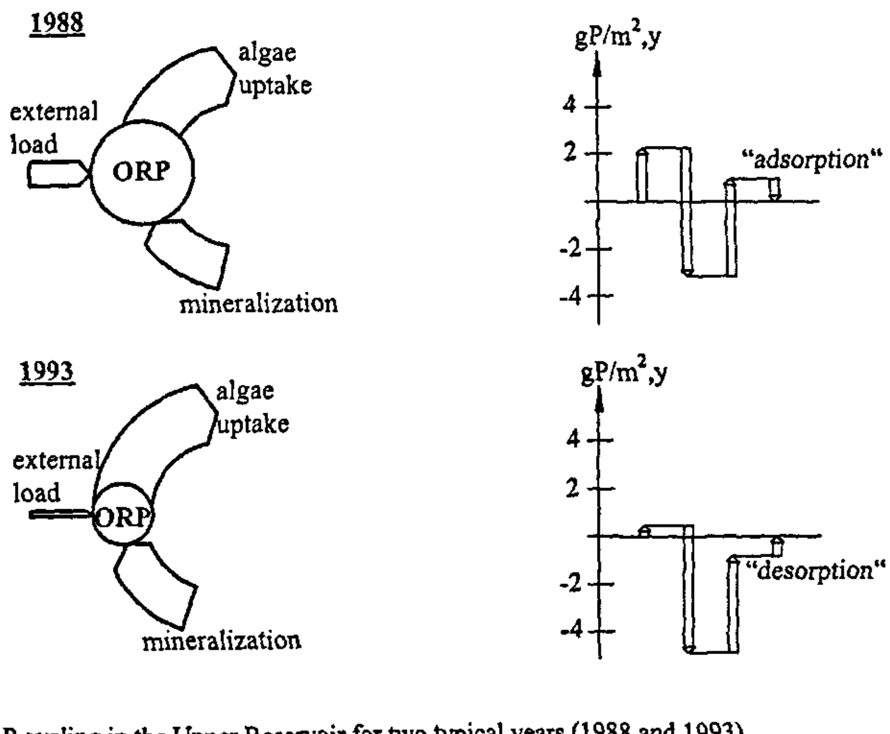


Fig. 4 P cycling in the Upper Reservoir for two typical years (1988 and 1993)

If modeling results were capable to capture impacts of external load changes, this would support our *broad "sorption" hypothesis* and would indicate the predictive power of the model for planning purposes of the Lower Reservoir. Without offering any details, the hypothesis also accounts for the difference in the open water and reed areas of the Kis-Balaton, which from the viewpoint of P removal, clearly suggests to favor the open water setting.

Models tested incorporated simple steady state TP ones and dynamic approaches (Clement et al., submitted). Steady state expressions (with the filling time or the apparent settling rate) were used with and without internal load. Model versions with internal load led to a Type 1 behavior supporting our hypothesis (Fig. 3 (b)), while all the others were unable to capture the sudden jump outlined earlier (Type 2, Fig. 3 (b)).

Dynamic state of the art P cycle models (similar to those developed for Lake Balaton, Somlyódy and van Straten, 1986) incorporated DRP, algae P, detritus P and inorganic particulate P as state variables (see Somlyódy et al., 1997, Clement et al., submitted). In Model I. of no memory effect, sorption was described by a simplified expression with an equilibrium concentration (DRP_e): $k(DRP - DRP_e)$ like in one of the earlier Lake Balaton models (van Straten, 1986). Model II. of a more complex sediment sub-model included DRP and particulate P in the active layer of the sediment (see later), as well as diffusion and adsorption/desorption (for the concept see the earlier explanation and Lijklema et al., 1986). Sorption was characterized by a three-parameter, so-called Koble-Corrigan isotherm of which the Langmuir expression is a special case (Somlyódy et al., 1997).

Parameter estimation by a Monte Carlo search method (Koncsos, 1994) was made by using TP, PO4-P and Chl-a as primary data and annual average outflow TP load as a secondary one. For details of calibration and validation the reader is referred to Somlyódy et al., 1997 and Clement et al. (submitted).

The internal load is illustrated in Fig. 2 (b) for the two-parameter modified Vollenweider model with constant apparent settling rate and Model I. The internal load starts to increase in 1990, it has a maximum in 1992/1993 which is followed - as anticipated - by a gradual decrease.

From the viewpoint of "macroscopic" behavior of the Upper Reservoir, dynamic Model I. (expressing the short term response of the reservoir) supported our hypothesis and corresponded favorably to Type 1 response, independently whether one, two or more spatial segments were applied (an estimate of the physical dispersion coefficient on the basis of the 2D hydrodynamic model indicated that 1-2 boxes should be sufficient, and thus at the beginning complete mixing was assumed). The usage of Model II. is discussed after outlining further assumptions on the behavior of soil and sediment.

Lower Reservoir

A soil survey was available for the area of both, the Upper and the Lower Reservoirs. However, the composition of soils and their nutrient balances were studied primarily from the viewpoint of agricultural production of limited use for limnological purposes.

Nutrient cycling of a freshly inundated reservoir is basically determined by soil conditions. With the formation of the sediment, the role of the original soil is decreasing. The rate and type of the change depends on many factors and an experimental study is hardly possible. As stressed before, this was the reason why observed alterations of the Upper Reservoir was utilized in a modeling framework.

In the Upper Reservoir, the water is oversaturated for calcite and significant amount of lime of high sorption capacity precipitates. On average, the rate of siltation is about 3 mm/y, 30 % - 40 % of which is due to $CaCO_3$ formation. The P adsorption capacity of the original soil gradually improves: by today the calcium content of the approximately 4 cm active layer increased from about 3 % of the original soil to 20 % or more and there is a clear evidence of the mixing of the sediment and the soil. The analysis of observations justified a sharp decline of the annual average DRP concentration in the water column as a function of the amount of $CaCO_3$ in the sediment (Istvánovics and Somlyódy, 1997). As a consequence of this, the cumulative sediment P concentration (in mg P/g $CaCO_3$ or in mgP/g dry sediment) and the above mean

DRP can be well characterized by a Freundlich isotherm, similarly to other shallow lakes (see Golterman, 1984).

All the above findings further justified our hypothesis and the use of a sorption isotherm for the sediment. The assumption that sediment and $CaCO_3$ formation will take place similarly in the Lower Reservoir (although at about a three times slower rate) led to the suggestion to transfer the identified isotherm to the Lower Reservoir, together with most of the parameters of Model II. (a few adjustments had to be made: for instance, due to the release of humic materials from the soil, the description of the extinction coefficient had to be modified).

The performance of Model II. thus obtained showed remarkable features. The fit to daily observations was excellent and a Type I. response looked for was resulted. However, the Monte Carlo calibration procedure led to a "surprising" isotherm with an inflection, as shown in Fig. 5. This could be hardly considered as a credible estimate.

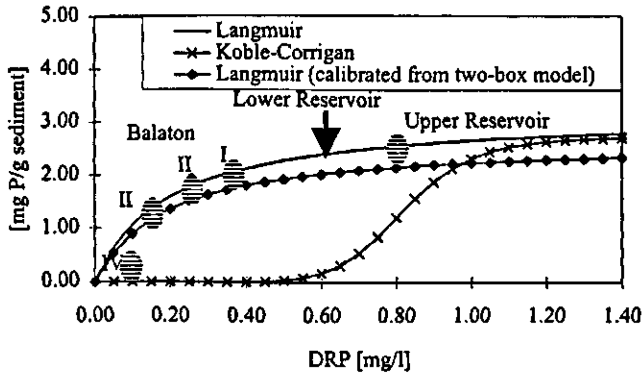


Fig. 5 Sorption isotherms

Next, long term simulations were performed jointly, for the Upper and Lower Reservoirs alike. Optimistic and pessimistic TP load scenarios (long term averages) were developed (the ratio of loads was about 1:3; see Somlyódy et al., 1997). The multi-annual variability of the TP loads as a function of the hydrologic regime was generated on the basis of the analyses of the daily Zala River observations. Since the load prior to 1992 was subject to control measures and other strong changes, for the purpose of the 16-year simulations, years 1992-1995 were selected randomly (with replacement). The procedure was also applied for hydrometeorological forcing functions. This allowed to capture not only the "trend" in the TP removal, but also its stochastic fluctuations typical for shallow waters.

Results are shown in Fig. 6 (a) and (c). As can be seen, the TP retention of the Upper Reservoir improves according to the model from about 30 % to 45 % due to the decrease of the internal load, while fluctuations are extremely large (particularly during the first ten years). The efficiency range of the Lower Reservoir is strikingly wide, there are lots of negative values during the entire simulation period and the trend-like efficiency improvement is marginal, only.

The figure is far from offering an optimistic view. Thus, recalling the Koble-Corrigan isotherm of Fig. 5, the dilemma is whether we trust our model calibrated and validated "nicely" to historical dynamic data sets. If the answer is "no", what can we do? In a little broader sense, how can we transform our task "to extrapolate from the past the future" into an interpolation?

As a first step we extended our *hypothesis* purely on the basis of intuition, and assumed that the formation of $CaCO_3$, the major sorbent, takes place "similarly" - following the hydrologic flow - from the Upper Reservoir to Basin IV of Lake Balaton (see Fig. 1), and the five different regions can be characterized by a single isotherm of no inflection point. The derivation of isotherm points were made on the basis of pore water measurements and sediment P concentrations calculated from mass balances - see Fig. 5 for the domains obtained and the Langmuir isotherm fitted.

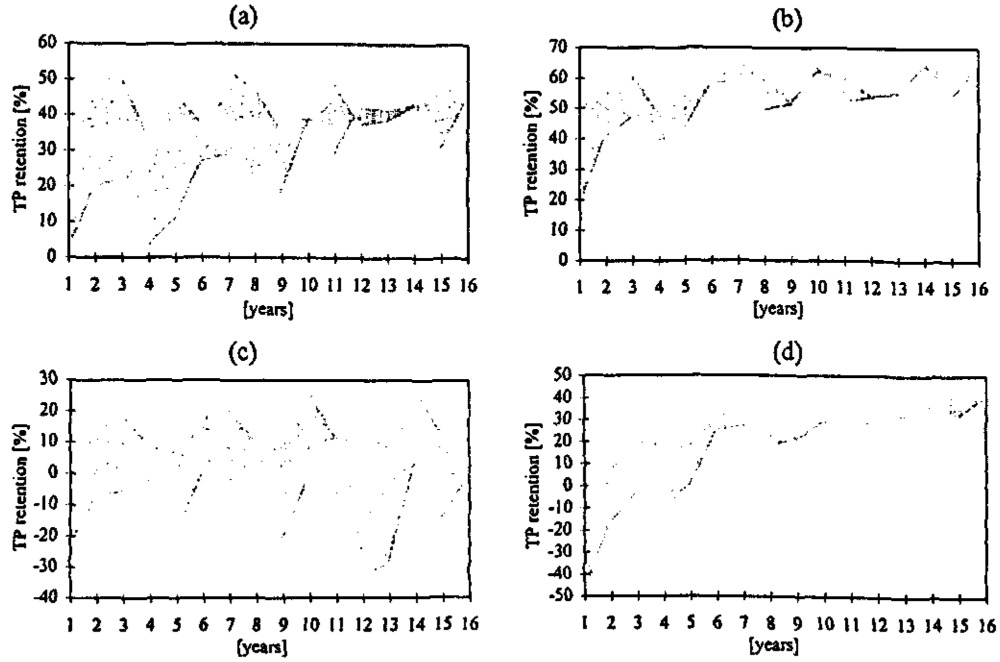


Fig. 6 TP removal of the Upper and Lower Reservoirs ((a) and (b), and (c) and (d), resp.) under various sorption assumptions and load scenarios ((a) and (c) - Koble-Corrigan isotherm; (b) and (d) Langmuir isotherm). Upper envelope - pessimistic TP load scenario; lower envelope - optimistic one

We seemed to be pleased with the long term simulations obtained. As apparent from Fig, 6 (b) and (d), efficiency ranges became significantly narrower, negative values do not occur after five years, and the system seems to reach an equilibrium within a decade or so. The annual average removal rate of the Lower Reservoir is slightly over 30 % (summer periods may be characterized by net P release) , which though is not an outstanding value, still it ensures under the optimistic scenario a TP load of the Keszthely bay close to the long term target. However, the serious drawback of this modification was that the dynamic performance of the model became poorer, and more importantly the model response was shifted from the preferred Type 1 to Type 2, which contradicted with our original hypothesis.

Next, we turned our attention to other possible causes of the above discrepancy. We addressed the question whether we did not overlook the role of "longitudinal" changes in the Upper Reservoir, particularly at the neck. It turned out that the distinction of the reservoir at least into two completely mixed boxes (without changing any parameter values) is a must (see Clement et al., submitted). Now, the Monte Carlo calibration resulted in a Langmuir isotherm very close to the previous one gained from our extended hypothesis, and independent data and calculations. The analyses led to results reflecting a harmony between fast dynamics of P in the water column and much slower changes in the sediment. TP retention rates remained practically unchanged as shown in Fig. 6 (b) and (d), and further insight was obtained on the longitudinal contribution of adsorption (being dominant in the neck area) and desorption (see Clement et al., submitted).

In spite of the positive outcome of the analysis, the reader should realize the large number of assumptions made, the justification of which is far from being complete. In fact, in many cases, like also here, we are simply missing methods to get a scientific evidence; operational experiences - if implementation will take place - are the only ways to learn how "inaccurate" we were.

RECOMMENDATIONS FOR POLICY MAKING

As mentioned, the problem of the Lower Kis-Balaton was a complex planning and decision making task of which P removal was just one element. The many-sided analysis of alternatives led to a proposal to modify the existing plan. Major recommendations were as follows (see Fig. 1): (i) To exclude the Ingói cops of unique value from P retention and flood protection, and to keep water level and its fluctuation solely according to needs of nature conservation; (ii) To use the rest of the reservoir for P retention and flood control; (iii) To inundate the reservoir quickly to shorten the transition period characterized by unavoidable die back of macrophytes; (iv) To upgrade the P removal of the municipal wastewater treatment plant of Keszthely to an effluent TP concentration of 0.5 mg/l and to divert the receiving channel to the upstream section of Area (III); and (v) To ensure under low flow conditions the temporary by-pass of the Zala River. This is advisable due to potential net P release of the reservoir during summers of the first few years of operation, the slow transition characterized also by decomposition of the accumulated organic matter and redistribution of the vegetation cover, as well as the guaranteeing overall flexibility of the operation.

In the course of the past nearly two years there were many policy oriented discussions and meetings on how to handle the Kis-Balaton issue. Sectors involved - water, environment and nature conservation - have gradually changed their original views and a dialogue have been developed. This led to the recognition that a compromise solution as outlined above should be looked for jointly and the implementation of the suggested modifications is likely. It was also realized that our scientific knowledge have improved a lot since the Kis-Balaton idea was addressed first (although still many gaps exist) and for today high resolution observations are available which help to understand better the behavior of the reservoir. Furthermore, in the meantime decision making became much more complex, due to our increased concern and sensitivity on environment and nature conservation issues, and the appearence of criteria associated.

SUMMARY AND CONCLUSIONS

Many advanced tools and models are available to support eutrophication management. Their use is often hindered by our limited knowledge: uncertain load-response relations typical for shallow, hypertrophic systems make difficult to "predict" the future. Additionally, eutrophication control is often just one element of a complex water resources project characterized by many conflicts and objectives changing with time. These features characterized our present example, the Kis-Balaton Reservoir system serving the control of nutrient loads of Lake Balaton. Detailed observations proved that P retention of the Upper Reservoir and the partially inundated Lower Reservoir functions differently then envisaged three decades ago and it also leads to the die off of a large, intact reed stand of high nature conversation value.

As a basis of the revision comprehensive analyses were performed to find a compromise solution satisfying different criteria (P retention, nature conservation, flood protection and others). As an element of the assessment, there was a strong focus on studying P cycling. Models of different complexity were calibrated and validated to the unique, daily observations of the Upper Reservoir. The model(s) obtained from the closest "case study" was then transferred to the Lower Reservoir to evaluate P retention for various alternatives.

The major dilemma was the description of the soil-sediment system and its transition which basically influences the development of the functioning of the reservoir. The validation of the model to daily data of the Upper Reservoir led to excellent dynamic performance, but the sorption isotherm obtained by a Monte Carlo estimation procedure and future predictions for the Lower Reservoir were considered far from being credible. In turn, a "credible" isotherm derived by using experimental data and intuition resulted in a poor historical response of the model. At the end, the incorporation of spatial changes by a multiple box model served reliable isotherm estimates, and a harmony between fast dynamics in the water column and much slower changes in the soil/sediment.

The problem of Kis-Balaton could be analyzed only by a modeling approach. However, the transformation of the overall task to "extrapolate from the past the future" into an interpolation can not be done without speculation and intuition. There are little, scientific grounds to justify our approach - only the implementation and operational experiences can show at what extent we were right or not.

ACKNOWLEDGMENTS

The study was financed by VIZITERV Ltd. We are indebted to the staff of the West Transdanubian Water Authority for their cooperation and particularly to P. Pomogyi for her many-sided, valuable assistance.

REFERENCES

Clement, A., Somlyódy, L. and Koncsos, L. (submitted) Modeling the Phosphorus Cycle of the Kis-Balaton Upper Reservoir, Water Science and Technology

Golterman, H. L. (1984) Sediments, modifying and equilibrating factors in the chemistry of freshwater. Verh. Internat. Verein. Limnol. 22, pp. 23-59

Herodek, S. and Somlyódy, L. (1996) Proposal to revise the design of the Lower Kis-Balaton Reservoir (in Hungarian), Manuscript, Budapest, p. 16

Istvánovics, V. and Somlyódy, L. (1997) Operational experiences of the Upper Kis-Balaton Reservoir (in Hungarian). Budapest University of Technology, Manuscript. Budapest, p. 62

Kárpáti, I., Kárpáti, V. and Pomogyi, P (1980) Study of the biological concept of the design and operation of the Kis-Balaton Reservoir (in Hungarian), manuscript, Keszthely, p. 144

Koncsos, L. (1994) The Model System and Software Package Reward (Regional Water Quality and Resources Decision Model), Manual (manuscript), Budapest, p. 163

Lijklema, L., Gelencsér, P., Szilágyi, F. and Somlyódy,L. (1986): Sediment and its Interaction with Water, pp. 115-182. In: Somlyódy, L., and van Straten, G., eds. (1986): Modeling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton. Springer Verlag, Berlin, p. 386

Novotny, V. and Olem, H. (1994) Water Quality. Prevention, Identification and Management of Diffuse Pollution, Van Norstand Reinhold, New York, p. 1054

OECD (1982) Eutrophication of Waters: Monitoring, Assessment and Control (Paris: OECD), p 154

Sas, H. (1990) Lake Restoration by Reduction of Nutrient Loading: Expectations, Experiences, Extrapolations, Vrh. Internat. Verein Limnol., 24, pp. 247-251.

Somlyódy, L. and van Straten, G., eds. (1986) Modeling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton, Springer Verlag, Heidelberg, p. 386

Somlyódy, L. and Wets,R.J.-B. (1988): Stochastic Optimization Models for Lake Eutrophication Management. Operations Research Vol.36, No.5. pp. 123-137

Somlyódy, L., Masini, I.A. and van Straten, G. (1989): Eutrophication of Lake Paranoá and Development of Brasilia, pp. 89-98. In: Salánki J. and Herodek S., eds. Conservation and Management of Lakes. Symp. Biol. Hung. 38. Akadémiai Kiadó, p. 423

Somlyódy, L. (submitted) Use of Optimization Models in Water Quality Planning, Water Science and Technology

Somlyódy, L., Herodek, S., Aradi, Cs., Clement, A., Istvánovics, V., Koncsos, L., Molnár, E., Rátky, I., Simonffy, Z., Szilágyi, F. and Várallyay, Gy. (1997) Revision of the design of the Kis-Balaton Lower Reservoir. Synthesis Report (in Hungarian), Budapest University of Technology, Manuscript, Budapest, p. 152.

Straskraba, M. (1995) Models for Reservoirs, Lakes and Wetlands, pp. 73-156, In: Novotny V. and Somlyódy L., eds., Remediation and Management of Degraded River Basins, NATO ASI Series, 2. Environment, Vol 3, Springer Verlag, Berlin, p. 529

Thomann, R. V. and Mueller, J. A. (1987) Principles of Surface Water Quality Modeling and Control. Harper and Row Publishers, New York, p. 644

van Straten, G. (1986) Hypothesis Testing and Parameter Uncertainty Analysis in Simple Phytoplankton-P Models, pp 287-308. In: Somlyódy, L., and van Straten, G., eds. (1986): Modeling and Managing Shallow Lake Eutrophication. With Application to Lake Balaton. Springer Verlag, Berlin, p. 386

THE EFFICIENCY AND JUSTIFIABILITY OF THIN-LAYER DREDGING IN THE KESZTHELY BAY (LAKE BALATON) IN ORDER TO REMOVE PHOSPHORUS

Vince Ördög and Elemér Dobolyi*

Pannon University of Agricultural Sciences, Institute of Crop Sciences,
Kolbai Károly u. 8, H-9200 Mosonmagyaróvár, Hungary

*Middle-Transdanubian Environmental Inspectorate,
Balatoni u. 6, H-8000 Székesfehérvár, Hungary

ABSTRACT

From the 4 basins of the shallow Lake Balaton, the Keszthely Bay is the smallest, but it receives the highest nutrient load (50% of the whole). Since the phosphorus sedimentation and accumulation is the highest in the Keszthely Bay, especially this part of the lake should be dredged in order to remove phosphorus. This argument, however, has divided the scientific community in Hungary into two parties. For this reason, the efficiency of the dredging have been investigated by chemical phosphorus fractionation of sediment samples, which were taken before and after dredging. Only few surface sediment samples (0-5 cm) have shown some improvement in chemical composition after the 15-25 cm thin-layer dredging. On the basis of the chemical data it was obvious that the dredging was rather harmful than useful. Since this establishment was not satisfactory, the bioavailable phosphorus was determined by bioassay. The dominant summer cyanobacterium species of the Lake Balaton, *Cylindrospermopsis raciborskii* was used as test-organism. In fact, after dredging the bioavailable phosphorus was less in the new surface sediment than in the old one. Consequently, less sediment phosphorus could support the phytoplankton biomass.

KEYWORDS

Bioassay, *Cylindrospermopsis racibroskii*, fractionation, Keszthely Bay, Lake Balaton, nutrient load, phosphorus.

INTRODUCTION

The Lake Balaton is the greatest shallow lake in Central Europe. Its main characteristics are the following: surface area = 596 km^2, length = 76 km, average width = 8 km, average depth = 3,25 m, volume of water at medium water level = 1861 million m^3. Its catchment area is 5774,5 km^2, 50-70% of the inflow being provided by the Zala river, the remaining 40% by smaller tributaries and periodical watercourses. From the 4 basins of the Lake Balaton, the Keszthely Bay is the smallest, but it receives the highest nutrient load through the Zala river (50% of the whole). From this we can conclude that it is Keszthely Bay where the phosphorus sedimentation and accumulation is the highest, i. e., it should be dredged in order to remove phosphorus and to decrease the phytoplankton biomass. This reasoning, however, has divided the scientific community in Hungary into two groups. Many seriously doubt the efficiency and justifiability of thin-layer dredging in this part of the Lake Balaton. Convincing results are necessary to resolve these doubts.

There are various methods to estimate the amount of phosphate releasing from lake sediment before and after dredging: (1) the phosphate exchange technique, (2) the chemical extraction technique, (3) the model approach and (4) the bioassay technique (Klapwijk *et al.*, 1982). From all of these methods the chemical extraction is the only one that doesn't take a long a time. On the other hand, the bioassay has the advantage of directly measuring the bioavailable phosphorus of the sediment (BAP). Research data show that the BAP in the upper 10 cm sediment layer of shallow lakes significantly contributes to the massive growth of phytoplankton (Tessenow, 1972; Lee, 1970).

In the present study undisturbed sediment core samples were collected before and after the thin-layer dredging of a selected area of Keszthely Bay. The samples were analysed by chemical fractionation, while the BAP was estimated by bioassay in order to determine the efficiency of thin-layer dredging.

METHODS

In the summer of 1996 undisturbed sediment core samples (0-30 cm) were collected with Eijkelkamp sampler in the Keszthely Bay before and after dredging in the summer of 1996. Ten samples had to be collected in equal distances from each other in the middle line of a 2500 x 500 m area. However, the selected area was not sampled at point 2, and there was no dredging done at point 10. The collected nine samples were cut into 5 cm long pieces and analysed by the generally accepted chemical fractionation method of Hieltjes and Lijklema (1980). Small portions of the samples were deep-frozen within 24 hours and stored in freezer (-20 °C) until the bioassay for the estimation of BAP. The results of the surface sediment layer (0-5 cm) will be presented and discussed in this paper.

Cylindrospermopsis raciborskii, a bloom-forming cyanobacterium of the Lake Balaton was used as the test organism in the bioassays. The strain was generously provided by Dr. Vörös from the Limnological Research Institute of Tihany (Hungary). The strain was maintained in BG-11 nutrient solution (Rippka *et al.*, 1979) deficient in nitrogen and containing only 10 % of the original micronutrients. Phosphorus-starved culture was prepared by growing of *C. raciborskii* during 7 days in modified BG-11 nutrient solution without phosphorus, and used as inoculum in the experiments. In the bioassays, the P-starved *C. raciborskii* was cultivated in BG-11 nutrient solution containing sediment as a sole P-source. The growth was determined by chlorophyll-a measurements in the seventh day of cultivation (Edler, 1979). The BAP-concentration was calculated on the basis of the calibration curve, which demonstrated the chlorophyll-a concentration of the P-starved *C. raciborskii* cultures in the presence of different PO_4-P concentrations. The bioassays were carried out in the apparatus described earlier by Ördög (1981). However, in order to meet the low light demand of the test organism the light intensity was decreased to 67 $\mu M.m^{-2}.s^{-1}$.

RESULTS AND DISCUSSION

The results of the chemical analysis clearly show that the values of the total P, HCl-extracted P, NH_4Cl-extracted P, and NaOH-extracted P (Fig. 1) are in many cases are higher after the dredging, than before the dredging. Consequently, according to these results the following conclusion should be drawn: the dredging is more harmful than useful! This conclusion, however is not acceptable either politically or scientifically. Politically it is not acceptable because the public expects some sort of action against increasing eutrophication, scientifically it is not acceptable because the continuously growing phosphorus load should somehow be reflected in the sediment.

Since we have failed to demonstrate by chemical analytical data after dredging an improved sediment quality, we have turned to the bioassay. The results of the bioassays proved the beneficial effect of the thin-layer dredging (Fig. 2). At six sampling points the BAP was significantly less after the dredging than before. There was only one sampling point, number 9, where an increased BAP concentration was measured. The average dry sediment, respectively.

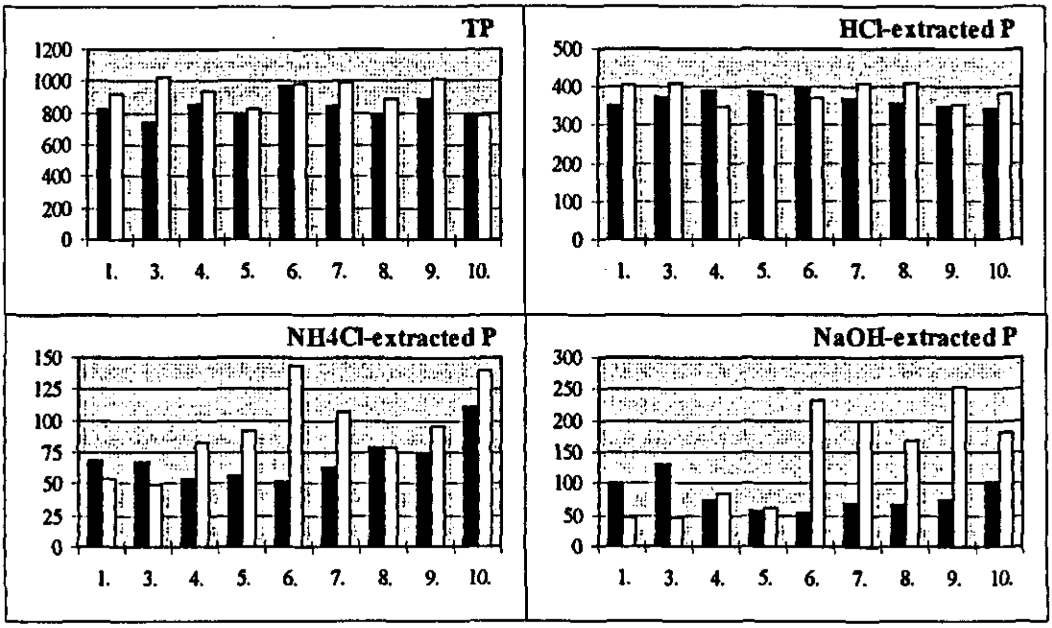


Fig. 1. Concentrations of the total phosphorus (TP) and the P-fractions extracted with different solvents according to Hieltjes and Lijklema in the surface sediment samples (0-5 cm) collected in Keszthely Bay of Lake Balaton. Horizontal axis: sediment samples, vertical axis: phosphorus in µg/g dry sediment, dark columns: P-values before dredging, white columns: P-values after dredging.

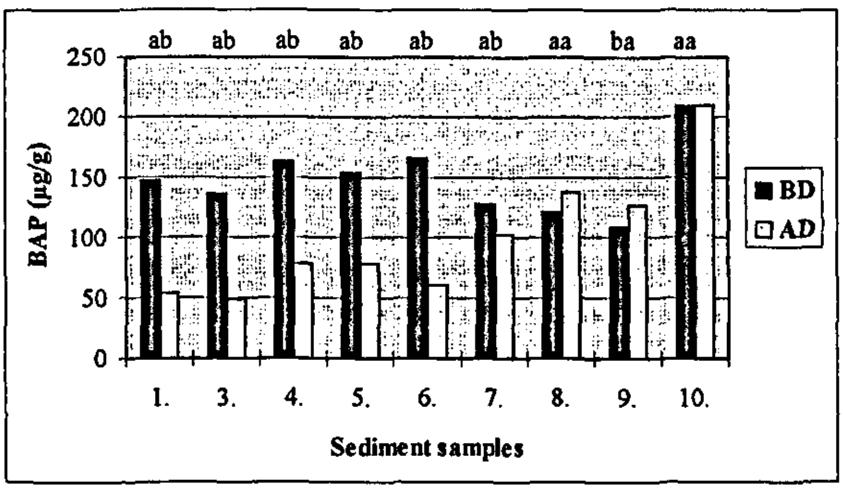


Fig. 2. Bioavailable phosphorus (BAP) concentration of the surface sediment samples (0-5 cm) collected in Keszthely Bay of Lake Balaton. The letters above the columns show the statistically significant differences (*ab* or *ba*) or similarities (*aa*).

average 17.9% before and 11% after the dredging. The decrease of the bioavailability of the sediment-P after the dredging was promising, but it could not be explained by the data of the chemical analysis. According to

Dorich *et al.* (1985) the suspended sediment could provide 28-41% algal available P, which might increase up to 80% (Hegemann *et al.*, 1983). Considering these research resulzs, the BAP in the sediment of Keszthely Bay is low, and it was actually decreased by the thin-layer dredging.

CONCLUSIONS

The results of the chemical analysis and the bioassays carried out with 9 sediment samples of Keszthely Bay confirm the following results:

1. The chemical phosphorus fractionation method was not suitable to determine the efficiency and justifiability of the dredging. Consequently, the chemical method could not be used for the selection of those parts of a lake sediment, which should be dredged in order to remove phosphorus effectively.
2. The bioassay, carried out with C. raciborskii, provided an effective tool for the above-mentioned purposes, and proved the efficiency of the thin-layer dredging.

REFERENCES

Dorich, R A, Nelson, D W and Sommers, L E (1985). Estimating algal available P in suspended sediments by chemical extraction. *J. Environ. Qual.*, **14**, 400-405.

Edler, L (1979). Recommendations on methods for marine biological studies in the Baltic Sea. Phytoplankton and chlorophyll. *The Baltic Marine Biologists*, **5**, 1-38.

Hegemann, D A, Johnson, A H and Keenana, J D (1983). Determination of algal-available phosphorus on soil and sediment: a review and analysis. *J. Environ. Qual.*, **12**, 12-16.

Hieltjes, A H M and Lijklema, L (1980). Fractionation of inorganic phosphates in calcareous sediments. *J. Environ. Qual.*, **9**, 405-407.

Klapwijk, S P, Kroon, J and Meijer, M L (1982). Available phosphorus in lake sediment in the Netherlands. *Hydrobiologia*, **92**, 491-500.

Lee, G F (1970). *Factors affecting the transfer of materials between water and sediments*. Univ. of Wisconsin Eutrophication Information Program, Literature Review No. 1., Madison, Wisconsin.

Ördög V (1981). Apparatus for laboratory algal bioassays. *Int. Revue ges. Hydrobiol.*, **67**, 127-136.

Rippka, R, Deruelles, J, Waterbury, J B, Herdman, M and Stanier, R Y (1979). Genetic assignments, strain histories and properties of pure cultures of cyanobacteria. J. Gen. Microbiol., **111**, 1-61.

Tessenow, U (1972). Lösungs-, Diffusions- und Sorptionsprozesse in der Oberschicht von Seesedimenten. I. Ein Langzeitexperiment unter aeroben und anaeroben Bedingungen in Fleibgleichgewicht. *Arch. Hydrobiol. Suppl.*, **38**, 353-398.

Lake Jamno Rehabilitation and Development Project, Poland

Hillebrand Ehrenburg, Iris de Jongh, Mario Maessen, Cees-Jan Vermulst

Grontmij Consulting Engineers, P.O.Box 203, 3730 AE de Bilt, the Netherlands

ABSTRACT

Pollution of large freshwater lakes greatly reduces their utilization potentials. A rehabilitation project for Lake Jamno is conducted by an association led by Grontmij Consulting Engineers in collaboration with DHV Consultants and Polish consultant NFEP. In this project the relevant technical, ecological and organizational aspects were studied. Lake Jamno is heavily eutrophicated. After an inventory of the available data an extensive water quality monitoring program was initiated. The monitoring program ended december 1996. Possible rehabilitation measures are reduction of external pollution loads, sediment removal and phosphorus inactivation. Different computer packages were used to predict the effects of possible measures. Water quality modelling was done using DUFLOW. Sediment transport was modelled using HISWA, DUCHESS and ESTRA. Because of the size of Lake Jamno and therefore the enormous costs of rehabilitation, a pilot project is proposed in which a compartmentalized part of the lake will be rehabilitated.

KEYWORDS

Eutrophication, lake rehabilitation, nutrients, sediment transport, water quality modelling.

INTRODUCTION

The catchment area of Lake Jamno (approx. 500 km^2) is situated around the Polish city of Koszalin, at the borders of the Baltic sea. The waters of Lake Jamno have experienced pollution problems since the past decades. This is an impediment to the development of the potential of Lake Jamno as a tourist recreation facility. The Lake Jamno Rehabilitation and Development Project aims at environmental rehabilitation of the lake and economic development of the area, mainly through tourism.

The Lake Jamno Rehabilitation and Development Project was designed as a result of a preparatory Masterplan for the restoration of Lake Jamno in 1993. The project is conducted by an association led by Grontmij Consulting Engineers in collaboration with DHV Consultants and Polish consultant National Foundation for Environmental Protection (NFEP). The project is financed by the Dutch Ministry of Economic Affairs (Senter).

PROJECT BACKGROUND AND OBJECTIVES

The Water Quality Management Project Lake Jamno started by the end of 1992. In the initial stages, besides activities concerning transfer of knowledge on water quality management between Dutch and Polish experts, the project focused on the provision of a technical, organizational, and financial basis for integrated water management in the catchment area. This resulted in a Masterplan for the area. The Union of Communes of Lake Jamno, vested as a result of the previous project phase, contains the municipality of Koszalin and the communes of Mielno, Będzino, Sianów, Biesiekierz, Manowo and Swieszyno.

The objectives of the ongoing project are to elaborate on the general long term development objectives for the catchment area as formulated in the Masterplan.
The general development objectives are:
1. improving environmental conditions in the catchment area, with a particular focus on improved water quality of Lake Jamno;
2. reinforcing the economic structure of the region;
3. coordinating and integrating institutional activities, which are required to support and stimulate initiatives to be undertaken by the Union in order for objectives 1 and 2 to be achieved.

Presently the project concentrates on drafting an environmental zoning plan, on drafting and executing an overall plan for waste water collection, transport and treatment and on institutional support for the Union of Communes.

This paper concentrates on remediation measures for the lake. The effects of these measures will first be studied with the help of a numerical model, comprising hydrographic, biological, chemical and diagenetic processes. Upon appraisal of the model results, the most promising measures should first be tested in a small-scale pilot study in a semi-confined part of the lake.
The mid-term aim of the project is to improve water quality, which today falls "out of class", to class-II according to Polish standards. This would enable the basic recreational functions to be resumed such as fishing, swimming and sailing. The project takes 30 months and started in January 1995.

RESULTS OF MONITORING AND PRESENT STATUS

From earlier Polish studies the general present status of the lake was inferred. The lake has a surface area of approximately 22,4 km^2 and averages 1.4 m in depth. During the past few decades water quality of the lake has gradually deteriorated as a result of increased uncontrolled discharges from domestic, urban, agricultural and, to a lesser extent, industrial wastes. In the seventies, significant quantities of organic matter (BOD) were observed. Today, the lake has become heavily eutrophic, visibility of the water is very low due to sediment resuspension and massive algae blooms and its water quality is poor, from a health, an ecological as well as from a recreational perspective.

During a monitoring and research programme for water quality, aquatic ecology and sediments these previous sources of information were extended and updated. The goals of the monitoring project were:
- to study the major biological and chemical processes operating in the lake;
- to assess a reference level for water quality, aquatic ecology and sediment quality;
- to identify the major problems of the lake and their underlying causes, in order to direct the choice of the most effective remediation measures;
- to gather the required input data (reference levels and processes) for a numerical model which is to be used to study the effects of the proposed measures.

From April to July 1995 and from February to November 1996, water quality measurements were carried out on a two-weekly or, for some components, monthly basis. Some twenty quality parameters were determined, such as pH, temperature, oxygen content, nutrients (see table 1), BOD and COD, major ions and coli counts. The results confirm the initial characteristics of highly eutrophicated waters of poor quality according to the Polish water quality standards for lakes with a recreation function. The present water quality, however, is not determined by micro pollutants, concentrations of which are invariably low for both organic and inorganic micro pollutants. The lake receives high nutrient input from the three main rivers. Uncontrolled domestic wastes, which posed the major problem in the past, will, however, drastically decrease in the near future as a result of structural improvements in the conditions of the sanitary infrastructure in the region.

Table 1. Mean water quality.

		Mean value [1]	target [1]
Total Phosphorus	(mg P/l)	0.42	≤ 0.10
Total Nitrogen [2]	(mg N/l)	4.11	≤ 1.50
Inorganic Nitrogen	(mg N/l)	0.80	≤ 0.40
Chlorophyll a	(μg/l)	145	≤ 15
Secchi depth	(meters)	0.35	> 0.50

1 Period: spring circulation and summer stagnation.
2 According to Kjeldahl method.

The distribution of water quality parameters shows little spatial variability, indicating the lake constitutes a well-mixed reservoir. About 1-5% of the water budget is derived from inflowing Baltic Sea water. Apart from external sources and hydrography, water quality parameters are intimately linked with biological processes, such as primary production, grazing and mineralization and, because of the latter, biogeochemical (diagenetic) processes in the sediment pore waters. Diagenetic processes seem to have a particularly strong influence on levels of oxygen and nutrients in the lake.

At the rise of temperature in spring, the development of a primary production bloom of diatoms is observed. During the bloom the nutrients that have accumulated during the previous period are consumed. When nitrate has become depleted, the diatoms are succeeded by green algae. By the end of spring the bloom collapses, probably due to a combination of nutrient limitation, limited light penetration and grazing. Grazing pressure, however, appears to be low. Numbers and biodiversity of the zooplankton are low. The reasons of this are presently poorly understood, but high turbidity and high predatory pressure by fish may contribute significantly. At the beginning of July the green algae are succeeded by a bloom of cyanophytes, leading to a noticeable decrease in water quality. Occasional patches of phosphorescent green water were observed in various parts of the lake.

By this time, mineralization of organic matter starts to draw heavily on the availability of oxygen, probably predominately in the sediments. Dissolved oxygen levels decrease, nitrate remains low and ammonia and phosphate levels strongly increase. The latter is due mainly to the reductive dissolution of iron-oxides in the upper sediment layers. Ammonia may partly derive from the incomplete mineralization of organic matter, but a more substantial fraction is believed to originate from the sediment pore waters. Especially during sulphate reduction, production of ammonia is high. The sulfides that are formed during this process, in turn, may get oxidized in the water column upon wind-induced mixing of the upper sediment. This would account for the observed increase of sulphate concentrations in the lake water as well as the increasing ratio of sulphate to chloride. This ratio cannot be accounted for by an increased contribution of river waters, as these contain too low sulphate concentrations and have too low ratios. Ratios in seawater are even lower.

The shift in inter-nutrient ratios (ammonia, nitrate and phosphate) on the one hand is determined by, and on the other hand determines the biological ecosystem composition. Ammonia is able to fuel a substantial regenerated production with high standing stocks of phytoplankton biomass. It is our intention to study these and other regulatory influences in bio-assay experiments.

In order to get a feeling for the exchange potential of nutrients from pore waters to the overlying water column, incubation experiments for phosphate were performed with undisturbed sediment cores. The results underline the enormous potential for dissolved phosphate fluxes from the

sediment. The incubation experiments will be continued, now extended with nitrogen components. This internal source is of comparable magnitude to that of external sources and may prove to be one of the main factors in the eutrophication problems.
In contrast, the sediment appears to be only slightly contaminated with micro pollutants. Concentrations of heavy metals are low, with the exception of moderately elevated mercury levels in a restricted part of the lake. Organic micropollutants are in mainly restricted to moderately elevated levels of PAH. Currently, no national Polish quality standard exist for fresh water sediments. There is, however, a standard for the quality of terrestrial soils, which imposes restrictions to the fate of any eutrophicated sediment that is to be removed from the lake. In order to put the pollution of the sediments into some perspective, the Dutch quality standard for sediments were adopted. This standard distinguishes five quality classes, ranging from 0 (clean) to 4 (heavily polluted). The sediments of Lake Jamno are classified as class 2 as determined by PAH and mercury, implying that they are moderately polluted with only minor actual or potential human and ecological risks. This is supported by analyses of fish, which do not show elevated contents of pollutants.

INTEGRATED APPROACH

The most important problems from an integrated water management point of view are the high degree of eutrophication, caused by high external loads as well as internal loads, the ensuing strongly impoverished biological diversity (not only of plankton but also of immersed macrophytes and zoobenthos) and the poor water quality. From this, albeit incomplete and preliminary, list of problems a selection of remediation measures was choosen:

- reduction of external loads by improving the sanitary infrastructure in the region (already in implementation);
- improving the quality of inflowing rivers by the construction of modern waste water treatment plants (a new plant for Koszalin has already started its operations);
- reducing the internal load.

Different computer packages were used to predict the effects of possible measures, comprising hydrographic, biological, chemical and diagenetic processes. Sediment transport was modelled using HISWA, DUCHESS and ESTRA. Water quality modelling was done using DUFLOW.
Upon appraisal of the model results, it is planned that the most promising measures will first be tested in small-scale pilot study in a semi-confined part of the lake.

REDUCING THE INTERNAL LOAD

To reduce the internal phosphorus source, the sediment of lake Jamno has to be rehabilitated. There are several possibilities which can be categorised in three groups:

- Sediment isolation
- Phosphorus inactivation
- Sediment removal

Sediment isolation is the least promising technique because Lake Jamno is too shallow for this type of measure. Phosphorus inactivation (Boers, 1990) is difficult because of the high resuspension rates in large shallow lakes. Sediment removal is in this case the most promising alternative because it is effective and irreversible. Jamno lake is very large so rehabilitation is expensive and time consuming. An alternative is to rehabilitate a small part of the lake. The most likely place to start is near Mielno. This area was chosen because of the already existing tourist accommodations and because of its shape (relatively easy to separate from the rest of the lake). This part can be separated by constructing an island, dam or vertical air/bubble screen. The nutrient rich sediment top layer (± 0.5 m) can be removed.

SEDIMENT TRANSPORT AND WATER QUALITY MODELLING

Sediment transport modelling

Before the internal nutrient loading of the South Western Corner of Lake Jamno is reduced by dredging, it has to be clear how much polluted sediment from the untreated compartment will be transported to the dredged compartment.

Wind induced waves cause resuspension, wind induced currents cause transport of suspended material. When wind action decreases, sedimentation will dominate over resuspension. Wind stress causes currents and waves. Waves are responsible for a circular movement of the water. The orbital velocities are able to bring bottom deposits in suspension. Wind induced currents are mainly responsible for transport of suspended material. Sediment transport was modelled with three numerical programmes from the Delft University of Technology: HISWA, DUCHESS and ESTRA.

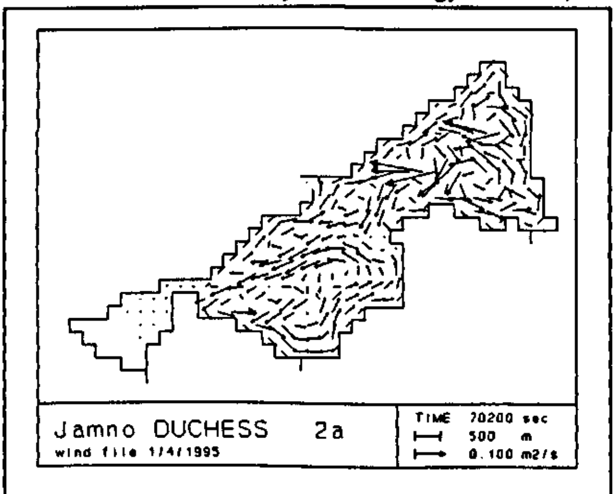


Figure 2: Dynamic flow field of suspended solids, induced by wave action and currents.

DUCHESS is intended to perform two dimensional flow computations. The program is based on a finite difference approximation of the two dimensional shallow water equations and the conservation of mass. The output is presented as a dynamic flow field.

HISWA is able to perform calculations of wave parameters in coastal areas from stationary wind-, bottom-, and current conditions. The output is given as a field of orbital velocities. These velocities cause resuspension of bottom sediments in Lake Jamno.

ESTRA is intended to perform two dimensional numerical calculations of transport due to tidal and wind driven flow. Transport of particles is described by conservation of mass and the advection diffusion equations.

The flow data calculated by DUCHESS and the orbital velocities calculated by HISWA are used as input for sediment transport calculations.

Water flow in the lake calculated with DUCHESS was used to calibrate the flow used in the water quality modelling with DUFLOW.

Water quality modelling

The numerical model was constructed using the DUFLOW program package. This package is designed for the simulation of one-dimensional unsteady flow in open channel systems. Although modelling Jamno lake is a two-dimensional problem, the one-dimensional model network was constructed in a way that the calculated water flow resembled the results of the two-dimensional model DUCHESS.

In the eutrophication type model the sediment top layer will be modelled too, which enables dynamic description of the fluxes across the water sediment interface. As the sediment acts as the memory of a system with respect to the loading history, this enables the model to be especially suitable for the simulation of longer time scales. Also in this model the cycling of phosphorus, nitrogen and oxygen is modelled. The model contains several algae species, which means that also the succession and the dynamics of the composition of the algal population can be simulated to a certain extend.

LAKE REHABILITATION SCENARIO'S

To gain the necessary understanding of the watersystem, 15 scenarios (table 3) were studied. The choice of scenarios was based on experiences in previous studies. By dredging alone, a nutrient reduction in the sediment of approximately 75 % can be established (Cooke, 1993). Dredging in combination with Iron Chloride injection can reduce the internal load up to 80%. Also the reduction of external nutrient input was reviewed. Two possible levels of external reduction were studied based on the input through point- and non point sources. Because wind has significant influence on the processes in the lake, average weekly wind velocities and directions are used as boundary conditions in the water quality calculations.

Table 3. Simulation scenarios of internal and external loading reduction.

External Reduction → Internal Reduction ↓	0% Phosphorus 0% Nitrogen	30% Phosphorus 20% Nitrogen	60% Phosphorus 40% Nitrogen
Dredge entire lake effectiveness 75 %	scenario 1	scenario 2	scenario 3
Dredge entire lake effectiveness 80 %	scenario 4	scenario 5	scenario 6
Dredge compartment effectiveness 75%	scenario 7	scenario 8	scenario 9
Dredge compartment effectiveness 80%	scenario 10	scenario 11	scenario 12
No Dredging	scenario 0	scenario 13	scenario 14

RESULTS

First a relatively rough simulation was made of all scenarios to compare the effectiveness of the measures on the water quality. The output of these simulations consisted of the development of nutrient and algae concentrations during a year. Figure 4 shows the development of anorganic phosphorus concentration in the lake in the calculated scenarios. This graph shows a maximum anorganic phosphorus concentration during the summer season at all scenarios. Depending on the scenario, the maximum calculated phosphate concentration varied from 0.22 mg P/l (bottom line in the graph) at scenario 6 (maximum internal and external reduction) to 0.50 mg P/l (top line graph) without reduction. Concentrations of other nutrients and algae showed a similar development.
Spacial differences in concentrations were negligible. Only in the compartmentalized scenarios concentration differences were detectable. The concentration differences depended strongly on the width of the narrow between the western part of the lake and the rest of the lake.
The calculated total nitrogen and total phosphorus concentrations in the 0 scenario (no treatment) are both below the measured values. This is caused by the used model, which does not calculate the effects of resuspension of organic matter. Since there was no data to calibrate resuspension, the calculations are carried out without resuspension.

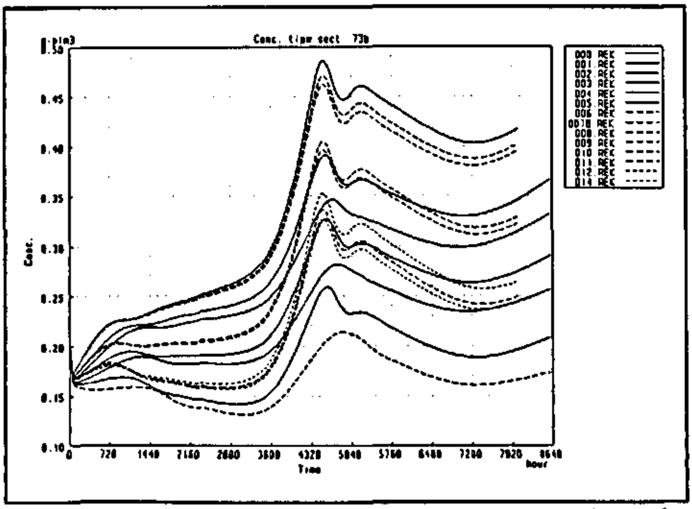


Figure 2: Total inorganic phosphorus concentrations of scenario 0 to 14.

The results showed that the reduction of internal loading is very effective. Without reduction of external loading, dredging the entire lake may result in a total nitrogen decrease from 2.5 to 1.0 mg N/l and total phosphorous decrease from 0.58 to 0.31. Reduction of external loading alone reduces the present concentrations to 2.0 and 0.37 respectively. A combination of these measures is most effective. The remaining phosphorus concentrations in all scenarios are still (far) above the target level. A combination of measures results in a total nitrogen concentration under the target level of 1.5 mg N/l, so reduction of nitrogen loading seems to be most effective in rehabilitating Lake Jamno.

Secondly, the present external loading in combination with dredging and iron application in a separated part of Lake Jamno was accurately simulated. These calculations were executed in order to get an idea of the impact on the lake water quality of the already accomplished external reductions in combination with internal loading reduction and to indicate if further external reductions are necessary. Due to dredging the total concentration of nitrogen in the sediment was decreased from 2 % to 0.5 %. Iron application does not effect the nitrogen concentration. The phosphorus concentration was decreased from 0.25 % to 0.05 %. Iron application inactivates phosphorus compounds in the sediment. This is simulated by decreasing the concentration of phosphate in the sediment to 0.025 %.

Table 3. Simulation of present external loading in combination with internal loading reduction.

	Present mean concentration in lake	Mean concentration after dredging	Mean concentration after dredging and iron application
Algae (mg C/l)	3.54	1.69	1.54
Total phosphate (mg P/l)	0.42	0.24	0.21
Total nitrogen (mg N/l)	4.11	1.18	1.18

The results (table 3) show that algae concentrations can be reduced by almost 60 %, which indicates that a chlorophyll concentration of about 90 μg/l may be achieved. The total nitrogen concentration is reduced to 0.44 mg N/l, which means that the target for total nitrogen can be reached by dredging. The phosphorus levels, however, maintain too high and indicate that further external phosphorus reductions are necessary. Especially since blue-green algae can take elementary nitrogen from the air for their growth, reduction of these algae must be obtained by phosphorus limitation.

CONCLUSIONS

- For successful rehabilitation of Lake Jamno both internal and external sources of pollution have to be reduced. The first step is the reduction of the external sources.
- Once the external sources of pollution have been reduced, the nutrient load in the sediments have to be reduced.
- The best method for reducing the internal nutrient load is removal of the sediments.
- If dredging has insufficient effect in reducing the internal nutrient load, Iron(III)Chloride injection can be used to increase phosphorus binding capacity of the lake sediment and so reduce the concentration in the water column.
- It is recommended to start a pilot project in a confined area in the S-W part of the lake.
- Removal of all polluted sediments is a prerequisite for restoration of the lake.
- The removed sediments can be stored in onshore and inlake depots. This would minimize costs and further create possibilities for shore- and island-related tourism development (yachting, windsurfing, swimming, angling).
- For the pilot project the sediments can be utilized to further narrow the opening between the two parts of the lake.
- The width of the remaining opening between the two parts of lake Jamno is very important and must be fairly narrow.
- To rehabilitate only part of the lake reduces the nutrient concentrations in that part, but the influence of the remaining part of the lake is relatively high. This means that the measure would be most effective if the compartment is shut of completely.
- Dredging the SW part of Lake Jamno causes an increase of sediment transport into this part both under N/NE and S/SW wind directions.
- Partly closing of the SW corner of Lake Jamno will probably lead to an increase of sediment transport per cross section and the sedimentation rate is estimated at approximately 1 mm/year because of:
  - small flow velocities (no sediment transport)
  - decrease of resuspension, increase of sedimentation (smaller fetch)
  - decrease of sediment transport out of the South Western corner of Lake Jamno
- When the south west corner has to be closed to prevent mixing, the opening has to be constructed in a way that flow in the opening is always east bound. This prevents mixing of nutrient rich water with water in the compartment, and inflow of suspended sediment.
- If Lake Jamno is compartmentalized the maximum width of the remaining opening between the compartments is less than 50 meters. The amount of transport of nutrients and suspended sediments through the remaining opening is strongly dependent on the width.

ACKNOWLEDGEMENTS

The authors are grateful to Delft University of Technology for its permissions to use the models DUCHESS, ESTRA and HISWA. We also would like to thank graduate student Martine van den Boomen, who has contributed considerably to the study.

REFERENCES

Cooke, G.D, Welch, E.B., Peterson, S.A., Newroth, P.R. (1993). *Restoration and Management of Lake and Reservoirs*, second edition, United States of America

P. Boers, 1990, phosphorus fixation with Iron(III)chloride: A New Method to Combat Internal Phosphorus Loading in Shallow Lakes, *Arch. Hydrobiol.* 129:339-351, Stuttgart, Germany

Trends in water quality and algae growth in shallow Frisian lakes, The Netherlands.

R. Maasdam and T.H.L. Claassen
Friesland Water Authority, P.O. Box 36, 8900 AA Leeuwarden, The Netherlands

ABSTRACT

About 20 shallow lakes form, together with interconnected canals, one hydrological water system in the province of Friesland. Total water surface area meets 14.000 ha, of which the lakes cover 10.000 ha. The regional catchment area is about 305.000 ha. In summer periods inflow of water from Lake IJsselmeer keeps the system on a constant water table of 0.52 m below sea level.

At the end of the sixtees the lakes became hypertrophic. Algae were dominating the biocenoses and submerged water plants disappeared. Systematic investigations started in 1970, and biological data were included from 1976 on. During twenty years a stable, but highly eutrophic situation, passed. Blue-green algae, especially *Oscillatoria agardhii*, were dominating the phytoplankton. Submerged vegetation was absent and bream dominated the fish-stock.

From 1991 on there is a slight improvement in water quality. Transparency increased, phosphate concentration and algae biomass decreased. Also a shift in phytoplankton species occurred. Dominance of *O. agardhii* decreased and species diversity increased. First submerged water plants recovered in 1994. Results of about 20 years water quality research is presented. Results are presentend as trends in time as well as within a gradient in the province. From south to north the chain-arranged lakes represent a remarkable pattern of increasing trophic state, caused by regional loading of nutrients.

KEYWORDS

Algae growth; eutrophication; gradients; *Oscillatoria agardhii*; shallow lakes; trends.

INTRODUCTION

In 1970 the Dutch Pollution of Surface Waters Act came into existence. For the inland waters the provinces are responsable for the policy of regional water management, water authorities are concerned with that management itself. So is the case in the province of Friesland. Water quantity management and water quality management of the Frisian lakes is practised by Friesland Water Authority. Thirty waste water treatment plants purify nearly all sewerage water. Monitoring of surface waters started already in 1960. However, that first -low frequent- measurements concerned only a few parameters like oxygen and chloride. From 1970 on nutrients were analysed and in 1976 hydrobiological investigations were incorporated in the monitoring network.

Water quality assessment occurs by comparing quality standards -as mentioned in the national Policy Plan of Water Management- with measured values. Table 1 presents standards for a few parameters related to the trophic state of surface waters.

Table 1. Water quality standards for a few selected parameters (Ministry of Transport and Public Works, 1989, 1997). Values for phosphate, nitrogen and chlorophyll-*a* as mean summer values.

		marginal values	indicative values
total phosphate	mg/l	< 0.15	< 0.08
total nitrogen	mg/l	< 2.2	< 1.5
chlorophyll-a	µg/l	< 100	< 50
transparency	m	> 0.4	> 0.5

For total phosphate a target value of 0.08mg/l has been set (Ministry of Transport and Public Works, 1997).

Fulfilling the national standards (table 1) should prevent negative consequences of eutrophication, such a mass algae blooms. However, indicative standards for a more natural situation are much lower. These standards (indicative values in table 1) are the same as formulated for the ecological objectives set for the basin water lakes (Grontmij, 1995). A much lower P-concentration should be realised (a standard of < 0.08 mg/l is mentioned) for preventing blue-green algae blooms (Ministry of Transport and Public Works, 1997).

In this paper results are presented of 14 years (1983 up to 1996) of water quality research in the Frisian basin water lakes. Variations between the lakes within the catchment area as well as trends in time will be presented. Special attention is given to nutrients (P and N) and algae (biomass and species composition), as well as there mutual relations. Because a water quality improvement became visible around 1991 a period before and after that date is worthmentioned. It is interesting to know whether this improvement is an ongoing proces. So that period is analysed and a short historical review is presented.

DESCRIPTION OF THE LAKES AND CATCHMENT AREA

The province of Friesland (313.200 ha on the main land) is situated in the northern Netherlands. In the eastern, middle and north-western part the soil consists of sand, peat and clay, respectively. The sandy area is above sea level (1 - 15 m), the peaty middle part just below sea level (down to -4 m), and the clayish area around or just above zero. Agriculture (mainly cattle-breeding) is the main activity in the rural areas.

The basin water system exist of lakes, interconnected by canals (Beattie *et al.*, 1978). The surface area is 14.000 ha, of which the lakes form 10.000 ha. The catchment-area meets 305.000 ha. Lake Tjeukemeer and Lake Fluessen/Heegermeer are the biggest ones, ca. 2100 ha each. Water level management in winter is characterized by discharging superfluous water to the Wadden Sea and in extreme (wet) situation also to Lake IJsselmeer. In summer, on the contrary, there is an inflow of water from Lake IJsselmeer the keep the water table on a constant level of 0.52 m below sea level. In table 2 a water balance, as well as P and N balances are presented for 1989-1990.

Table 2 . Global water (in million m^3), phosphate and nitrogen (in 1000 kg) balances of the Frisian basin water system for 1989-1990 (after Raad *et al.*, 1993).

	water		P		N	
	winter	summer	winter	summer	winter	summer
IN						
Precipitation/Deposition	55	57	9	9	290	298
Inflow from Lake IJsselmeer	4	187	0	30	15	722
Waste water/sewage	45	44	115	122	742	787
Run off from polder-areas	525	86	320	50	8760	1200
In total	629	374	444	211	9807	3007
OUT						
Evaporation/Internal losses	16	66	142	57	5205	1546
Discharge to Groningen	5	97	2	41	23	374
Outflow Lake Lauwersmeer	448	132	221	70	3379	680
Outflow Harlingen	46	79	23	43	341	407
Outflow Lake IJsselmeer	114	0	56	0	859	0
Out total	629	374	444	211	9807	3007

As seen in table 2 the inflow of Lake IJsselmeer water in summer amounts 50 % of the water balance. However, for phosphate and nitrogen these percentages for the summer period are only 14 % and 24 %, respectively. Discharges from polder areas are by far the greatest source of P as well as N, with respectively 56 % and 78 % on an annual basis. By realising phosphate removal on all treatment plants this amount is already reduced at the moment.

Beside Lake IJsselmeer the 16 most important basin water lakes are included in the figures 2 up to 5. In table 3 these lakes are mentioned, while locations are indicated in figure 1. Characteristic is the phenomenon of a chain-arranged lakes system througout the catchment area. The "hydrological distance" is the distance measured from the sampling station to Lake Ijsselmeer over water.

Table 3. The lakes in study, with some characteristics (partly after Beattie *et al.*, 1978; Claassen, 1986).

sampling station	lake	surface area (10^6 m2)	average depth (m)	volume (10^6 m3)	hydrological distance (km)
121	Grote Brekken	2.9	1.5	4.6	4.2
89	Koevorder	3.4	1.0	3.7	12.1
90	Langweerder Wielen	2.1	0.9	1.9	16.0
75	Sneekermeer	12.2	1.5	14.6	22.9
120	Brandemeer	0.5	1.3	0.7	7.5
105	Slotermeer	11.2	1.6	17.0	12.4
142	Tjeukemeer	21.5	1.7	35.7	10.9
85	Fluessen	15.8	1.8	28.4	14.0
86	Heegermeer	7.0	1.5	10.5	18.6
53	Pikmeer	0.8	1.4	1.1	32.5
54	Sitebuurster Ee	1.5	1.5	2.3	35.7
51	Alde Feanen	4.5	1.2	5.4	39.4
56	Wijde Ee	1.1	1.3	1.4	41.6
57	Smalle Eesterzanding	0.4	0.7	0.3	45.6
45	de Leijen	2.9	1.4	3.9	54.1
34	Bergumermeer	4.3	1.3	5.7	51.7

MONITORING PROGRAM

As mentioned before water quality monitoring started already in 1960. From 1970 the lakes were incorporated in the monitoring network. Here results from 1983 on will be presented. Sampling stations in the lakes as well as along the main axis from Lake IJsselmeer (southern part of the province) to Bergumermeer (northern part) have been selected for presentations. In figure 1 the situation of the lakes in study are presented, some characteristics are presented in table 3.

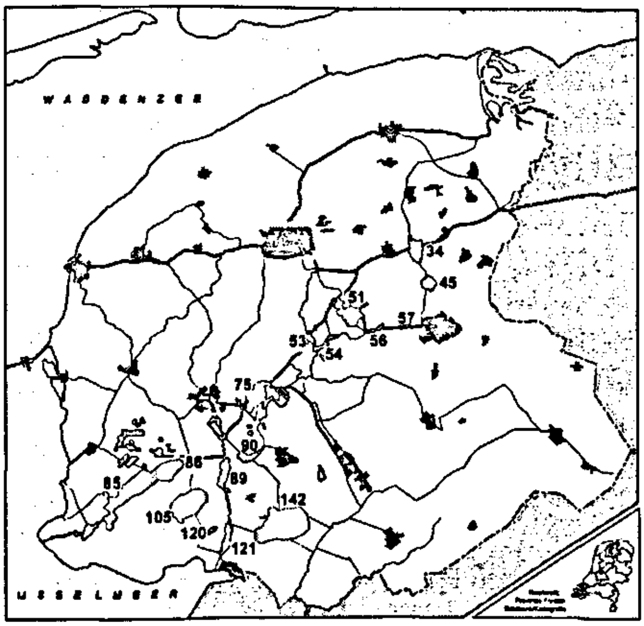


Figure 1. The province of Friesland with the basin water system and the whole catchment area and the sampled lakes (see also table 3 for sampling points).

The monitoring program included monthly physico-chemical analysis of 20 parameters, listed as parameters of the basic water quality standards. Additionally phytoplankton composition (one, three or six times a year) and water plants were investigated too (Friesland Water Authority, 1993, 1994, 1995, 1996).

RESULTS AND DISCUSSION

Geographical gradients. In the seventies and eighties the (hyper)trophic state was rather stable with mass algae growth, in summer dominated by the blue-green *Oscillatoria agardhii*. In figure 2 the situation for August 1983 is presented. Most lakes were dominated by *O. agardhii*, and the phytoplankton diversity was poor (Claassen, 1986). Water plants were still absent. This pattern was found -only with small yearly variations- up to 1990. From that time an improvement in water quality to a less eutrophic situation became visible. In figure 2 that improvement is visualisized. The maximum levels of *O.agardhii* decreased to 60 % and 20 % in 1993 and 1996 respectively. At the same time the percentages of green algae (Chlorophyta) as well as the number of taxa increased.

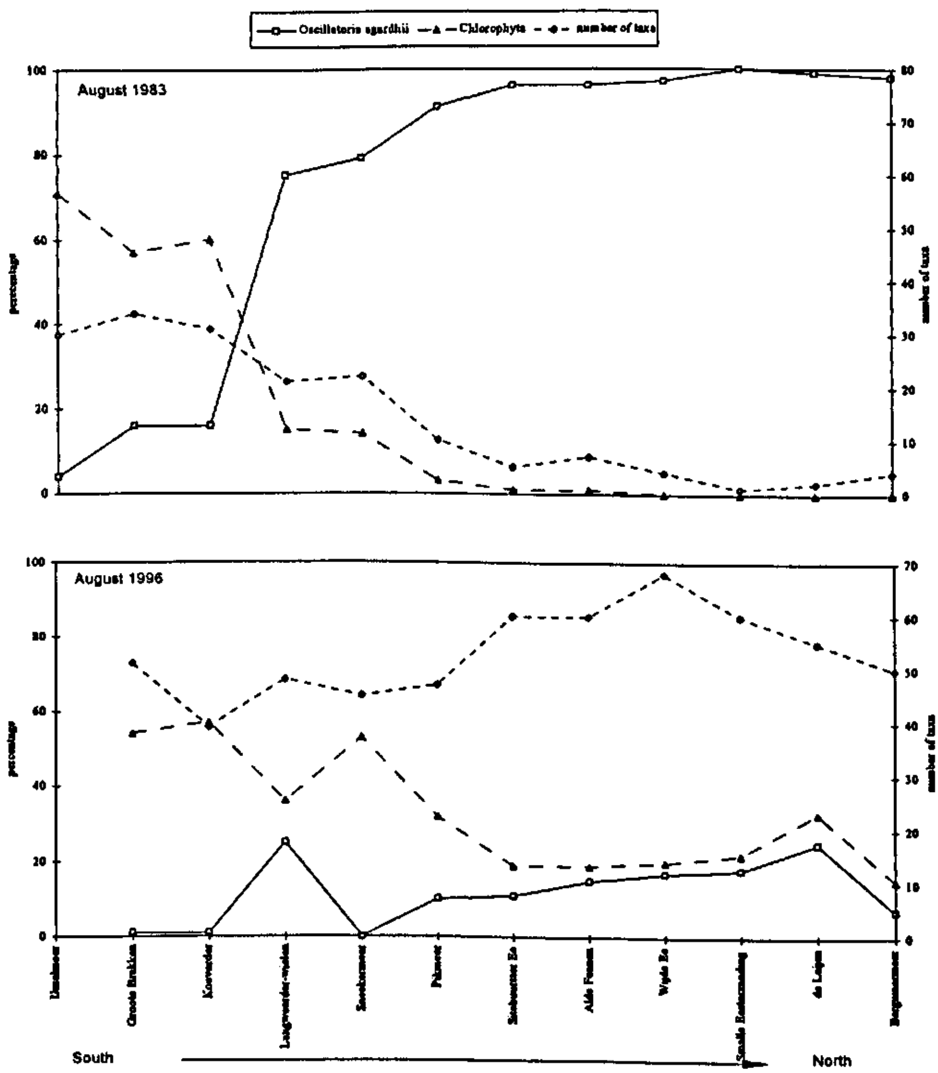


Figure 2. Percentages of *Oscillatoria agardhii* and Chlorophyta, as well as the number of phytoplankton-taxa present in August samples.

From south (Lake IJsselmeer) to north (Lake Bergumermeer) there is a remarkable gradient in values for a number of water quality parameters, as seen in figures 2 and 3. In figure 3 the course of the P- and N-concentrations are given, as mean summer values and as ranges of concentrations througout the year. There is an increase in P concentrations up to the north, mainly caused by internal discharges of effluents of treatment plants and by discharges of superfluous polder water on the basin water system (De Haan & Moed, 1984; Boers, 1996), see also table 2 for P and N balances. For N the level is high, but rather constant probably caused by an equilibrium between discharges and denitriphication. Even the minimun concentrations are as high as the (mean) standards for both nutrients.

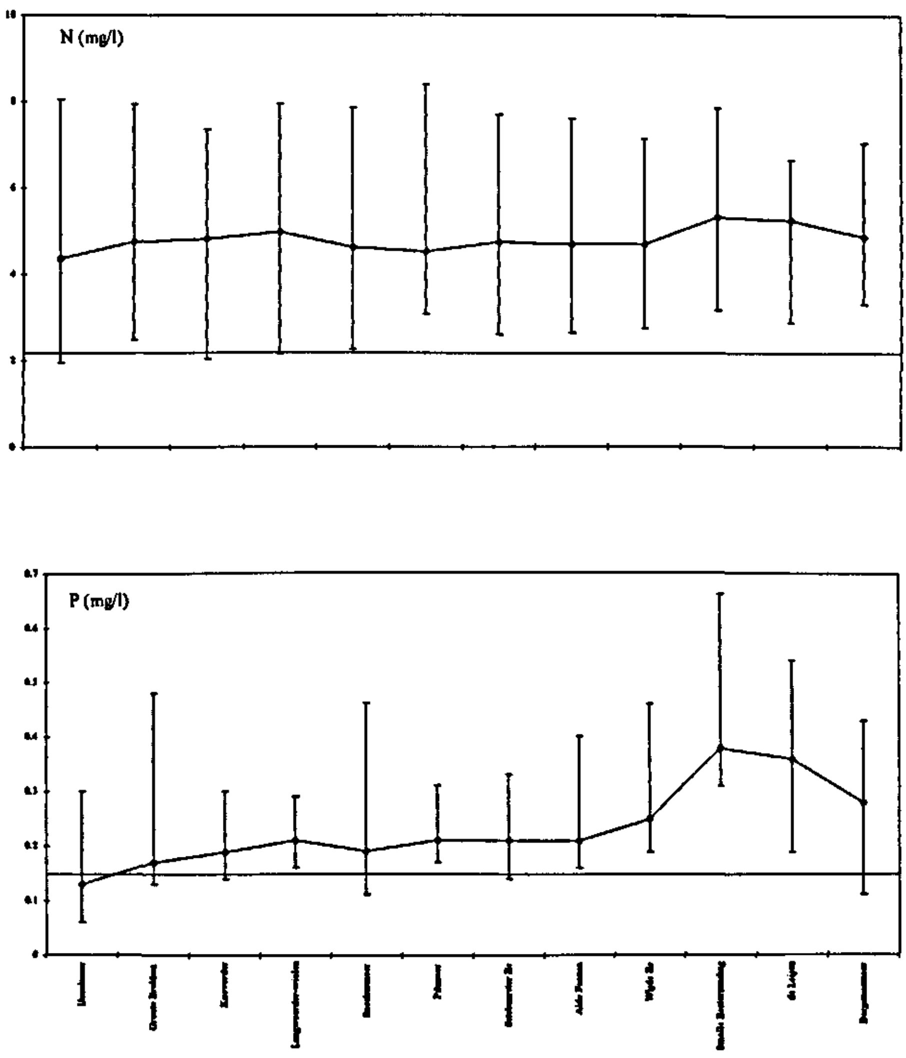


Figure 3. Phosphate and nitrogen concentrations for 1995 in the south-north gradient from Lake IJsselmeer to Lake Bergumermeer, related to the national standards of 0.15 mg/l and 2.2 mg/l, respectively.

Trends in time. In figure 4 the percentages of the sampling stations in the basin water system, which fulfil the standards for phosphate, nitrogen, chlorophyll-*a* and transparency (see table 1) are presented for the period 1984-1996. Nearly all stations have mean N concentrations higher than 2.2 mg/l. However, a remarkable improvement took place for chlorophyll-*a* in 1991, for transparency in 1992 and for phosphate (although still fluctuating from year to year) in 1993.

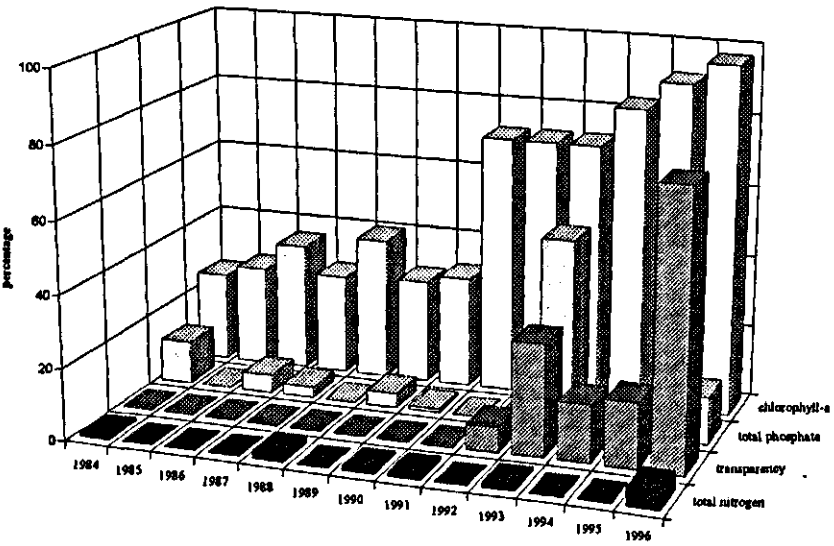


Figure 4. The percentages of sampling stations in the basin water fulfilling the national standards (marginal values) as given in table 1 from 1984 up to 1996.

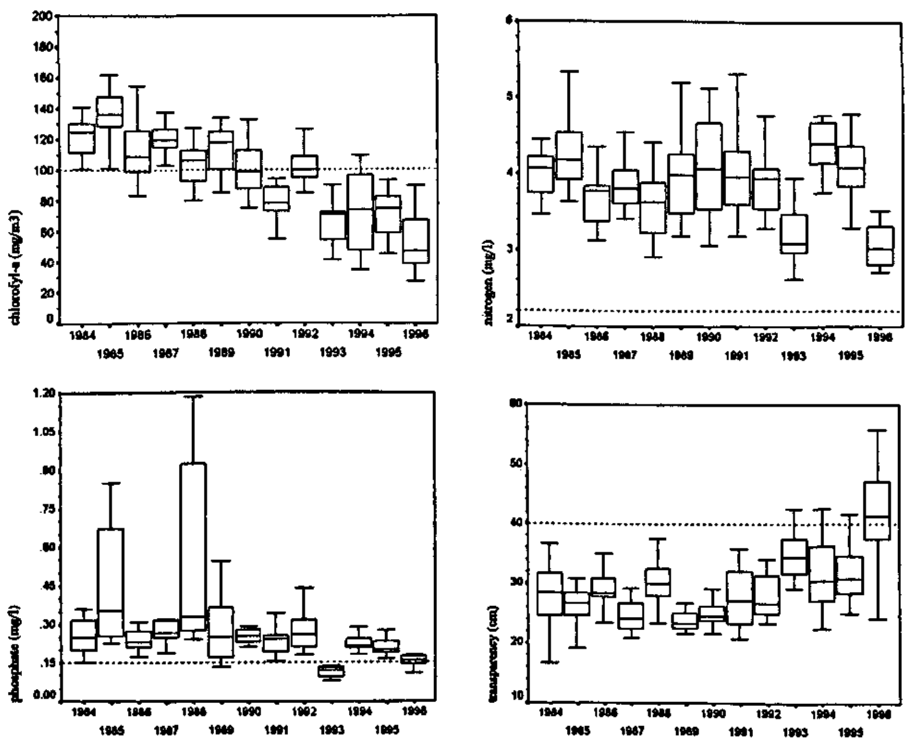


Figure 5. Boxplots of mean summer values for 16 lakes (see table 3) for the parameters chlorophyll-*a*, nitrogen, phosphate and transparency.

In the period up to 1990 water quality in the lakes was constantly hypertrophic with little variation annually. A slight improvement is visible from 1991 on. Most striking is the number of sampling station fulfilling the standard of < 100 µg/l for chlorophyll-*a*. Transparency and phosphate show also some improvement. However, nitrogen concentrations did not decrease at all. Box-whisker plots for the overall summer mean values of the 16 lakes (see table 3) are given in figure 5 for the period 1984 up to 1996. Phosphate values show a slight decrease in average. However, the maximun values decrease more pronounced. The years 1993 and 1996 show rather low concentrations, mainly caused by preceeding of relatively dry winter periods and as a consequence less run off from polder areas. For nitrogen both years are equally low, but in the other years there is no trend at all. Beside all values are far beyond the standard of 2.2 mg/l. The chlorophyll-*a* concentraties within the lakes show a gradually decrease and from 1991 (except 1992) meet the standard of 100 µg/l. Parallel the values for transparency are increasing and, although still rather low, are reaching the standard of 0.4 meter.

CONCLUSIONS

- The shallow Frisian chain-arranged lakes are typically for the northern Netherlands, and a good in-situ study-object;
- The lakes are hypertrophic form the sixties: algae dominate the biocenoses and macrophytes disappearred (between 1960 and 1970);
- There is a slight but constant improvement from 1991 on,

indicated by:

- changes in algae composition with a shift from blue-green to more green algae, return of some water plants, a lower fish-stock of bream, an increase of zebra mussels, clearer waters for short moments and/ or specific locations;

and caused by:

- a number of restoration measures and management tools have contributed to this slight improvement, as there are phosphate removal on waste water treatment plants, biomanipulation by removing bream in the period 1989/'90 - 1993/'94, reduction in agricultural uses of manure especially in winter, improving of the water quality of Lake IJsselmeer water induced by national and international agreements. This improvement of trophic status is much more regulated by P (reduction) as by N. As a result of this less eutrophic condition there is an influence of climatological conditions (*e.g.* in 1993, 1996) on water quality. Although improvements in water quality the aquatic ecosystem is not yet in an equilibrium of a clear water phase.
- Ongoing measures for further improvement the eutrophic situation is still urgently needed. Further actions are necessary for passing the turning point from turbid water to clear water lakes. The marginal values of trophic water quality standards are not yet reached, even less is the case with indicative values (see table 1). Continuing monitoring (also with other technics like remote sensing (Claassen *et al.*, 1994), bioassays (Bolier, 1994) and modelling (Brinkman *et al.*, 1988) is necessary for an optimal and effective water management.

REFERENCES

Beattie, D.M., Golterman, H.L. and Vijverberg, J. (1978). An introduction to the limnology of the Frisian lakes. *Hydrobiologia* 58(1), 49-64.

Boers, P.C.M. (1996). Nutrient emissions from agriculture in The Netherlands, causes and remedies. *Wat. Sci. Tech.* 33(4-5), 183-189.

Bolier, G. (1994). Criteria used to assess nutrient limitation of algae by bioassays. *Verh. Internat. Verein. Limnol.* 25, 1917-1921.

Brinkman, J.J., Groot S., Los, F.J. & Griffioen, P. (1988). An integrated water quality- and water quantity model as a tool for water management; application to the province of Friesland, the Netherlands. *Verh. Internat. Verein. Limnol.* 23, 1488-1494.

Claassen, T.H.L. (1986). The Frisian lakes as a shallow hypertrophic *Oscillatoria*-dominated system. H_2O 19(12), 268-275, 279.

Claassen, T.H.L., Buiteveld, H., Roeters, P.B. & Appelman, K. (1994). Eutrophication research by remote sensing, applied in Friesland. H_2O 27 (25), 740-745.

Friesland Water Authority (1993,1994,1995,1996). Quality of surface waters in Fiesland 1992, 1993, 1994 and 1995.

Grontmij (1995). Ecological Management Programme for Basin water lakes in Friesland. Zeist. 128pp. with

supplements (in Dutch).
Haan, H. de and Moed, J.R. (1984). Phosphorus, nitrogen and chlorophyll-*a* concentrations in a typical Dutch polder lake, Tjeukemeer, in relation to its water regime between 1968 and 1982. *Wat. Sci. Tech.* 17(2), 733-743.
Hosper, S.H. (1984). Restoration of lake Veluwe, The Netherlands, by reduction of phosphorus loading and flushing. *Wat. Sci. Tech.* 17(2), 757-768.
Ministry of Transport and Public Works (1989). Eutrophication; document of the Third Policy Plan of Water Management. RWS/RIZA. 71pp. (in Dutch).
Ministry of Transport and Public Works (1997). Fourth Policy Plan of Water Management. (in Dutch).
Raad, J.S., Peerboom, R.A.L. & Feenstra, J.F. (1993). Balances of the Frisian basin water system. IVM, Amsterdam. 110 pp. (in Dutch).

RESTORATION OF THE LAKE NANNEWIJD: FIRST RESULTS

R.Veeningen
Water Authority Friesland, PO Box 36, 8900 AA Leeuwarden, the Netherlands

ABSTRACT

In lake Nannewijd, a small polder lake in Friesland, a number of restoration measures have been realised in the period April 1993 up to April 1995. Each separate measure contributed to the improvement of the water quality of the lake.

KEY WORDS
eutrophication, restoration, water quality, hydrological isolation, reed filter, dredging, phosphate fixation, biomanipulation.

INTRODUCTION

Lake Nannewijd is a small and shallow (area, 100ha; mean depth 1m) lake in the peaty middle part of the province of Friesland. Submerged water plants are absent and phytoplankton dominates the biocoenoses. The lake has a function for recreation and nature conservation. The water quality problems of lake Nannewijd relate to: green colour and high turbidity, a thick layer of soft, nutrient-rich sediment, an insufficient inlet of water in summer and outflow in winter of water through the lake. The drainage of the agricultural polder is poor and fishdiversity is low. The restoration measures were carried out from April 1993 up to April 1995.

MEASURES

The measures are outlined below. Numbers in the text refer to locations in Fig. 1.

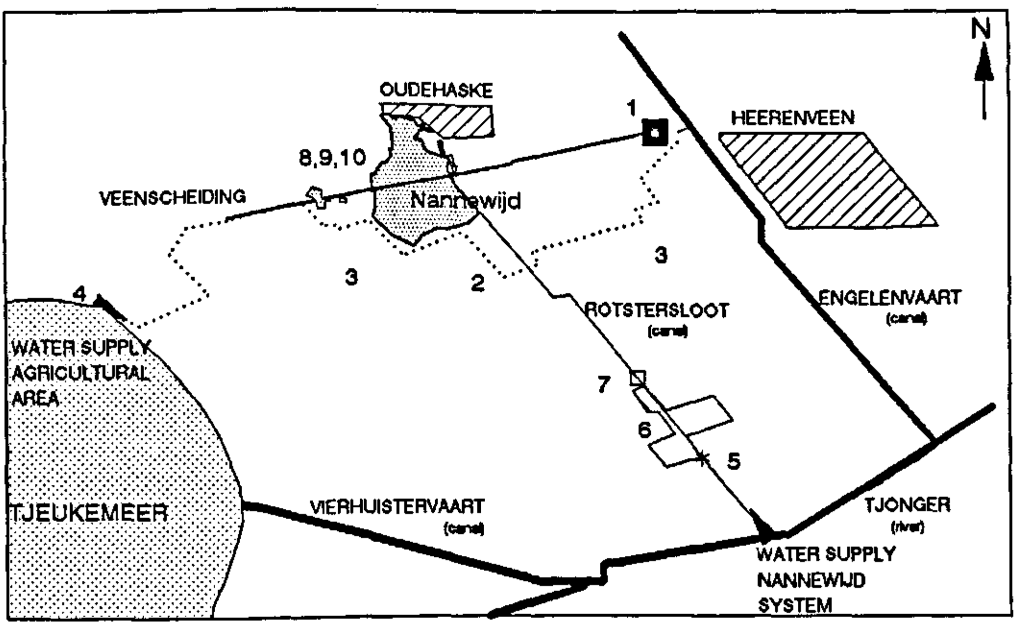


Fig. 1. Measures ont the map.

Hydrological isolation

- the lake is hydrologically isolated from the agricultural surroundings (2);
- the lake gets its supplementary water from the river Tjonger (5-6-7);
- the agricultural area gets its water from the lake Tjeukemeer (4);
- the surplus water from both the Nannewijd-system and the agricultural watersystem is pumped out by a renewed pumping engine 'de Foarutgong' (1);

The reed filter and the chemical treatment (5-6-7)

- the water for the lake is pumped from the Rotstersloot (5) into the reed filter (6) and afterwards the effluent is treated chemically for P-removal (7);
- the net surface of the reed filter is 18 ha (formerly used as agriculture area);
- reed was grown from seeds in glass houses and afterwards transplanted to the filter area;
- the plants were planted by a crop planting machine within 3 weeks;
- the average density of reed was 3 plants per square meter;
- while the average concentration of total-phosphorus is about 0.15 mg P/l, iron-chloride is added at a concentration level of about 15 mg ferric iron per litre;
- the iron-phosphate settles in the canal (Rotstersloot) to the lake.

Sediment removal (8)

- about 100.000 m3 of sediment was removed from the lake and deposited at two places near the lake (the deposits could be used for agricultural purposes after drying);
- sediment was removed by dredging in only 25 % of the total area: in the rest the sludge layer was thinner than 10 cm;
- after sediment removal about 70.000 m3 nutrient rich sludge is still present in the lake.

Phosphate fixation (9)

- to prevent phosphorus-release from the sediment after restoration, the sediment was treated with ferric-chloride;
- it was ensured using a Digital Geographical Position System that every square meter was treated with iron-chloride;
- the amount of iron-chloride increased with the depth of the sediment-layer: 0-5 cm , 0,13 l $FeCl_3$ per square meter; 5-10 cm, 0,39 l $FeCl_3$ per square meter; and > 10 cm, 0,65 l $FeCl_3$ per square meter;
- $Ca(OH)_2$ was added to prevent acidification of because of the low buffering capacity).

Biomanipulation (10)

- the fish population in the lake was supposed to be dominated by bream;
- the goal of the biomanipulation was to remove bream and carp standing crop for more than 80 %;
- to control the biomanipulation at first 2000 specimen of bream were removed, marked by a small cut and reintroduced in to the lake; by doing so 83 % of the fish stock was reduced;
- in total about 14.000 kg's of small and large breams were removed;
- the population of carp was smaller than expected.

RESULTS AND DISCUSSION

Hydrology

The hydrology of the lake has been drastically changed by the hydrological isolation (Fig. 2) The residence time in summer is prolonged stronghly.

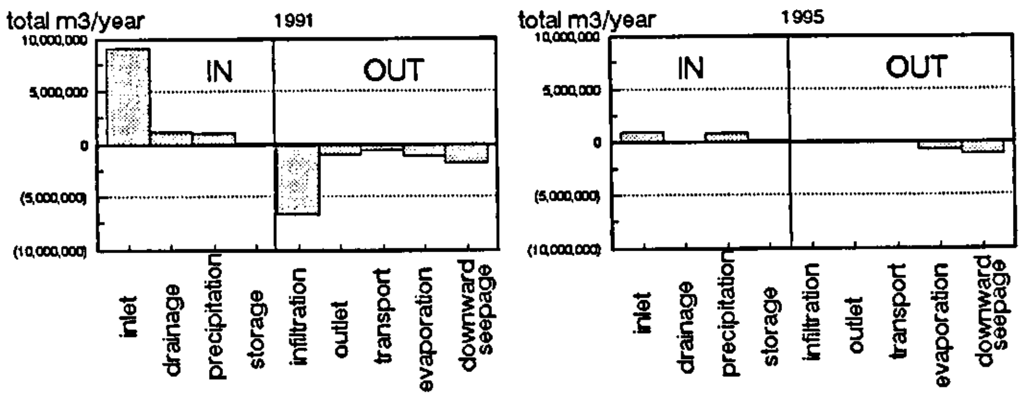


Fig. 2. The waterbalance of the lake before and after the measures.

Reed filter

The supplementary water used was totally devoid of suspended matter (Fig.3.). The efficiency for removal of phosphorus and nitrogen is less than that for suspended matter. Concentrations of nitrogen and phosphate after passing the reed filter are quiet similar to those for the water quality standards (0.15 mg P/l and 2.2 mg N/l). The chemical treatment contributes to the reduction of phosphorus in case of high concentrations in the effluent of the reed filter only.

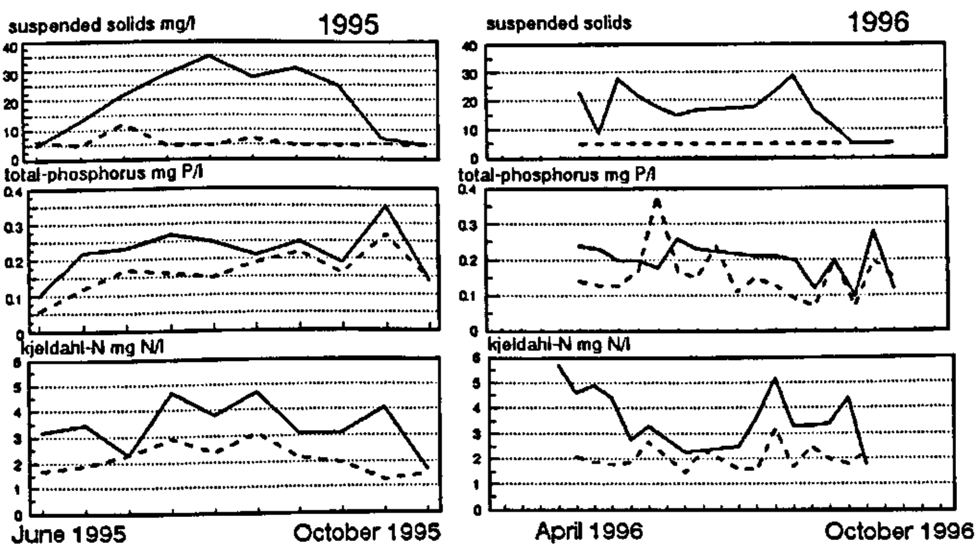


Fig. 3. Water quality parameters before (solid line) and after passing (dashed line) the reed filter.

Dredging and phosphate fixation

Due to technical problems (it was only the second time that this method was applied in the Netherlands) it took 5 months to treat the whole lake. From Fig. 4. it is clear that phosphorus release is reduced after the dredging and sediment removal and the phosphate fixation. Although there was a decline of the pH on phosphate fixation, it dit not became critical.

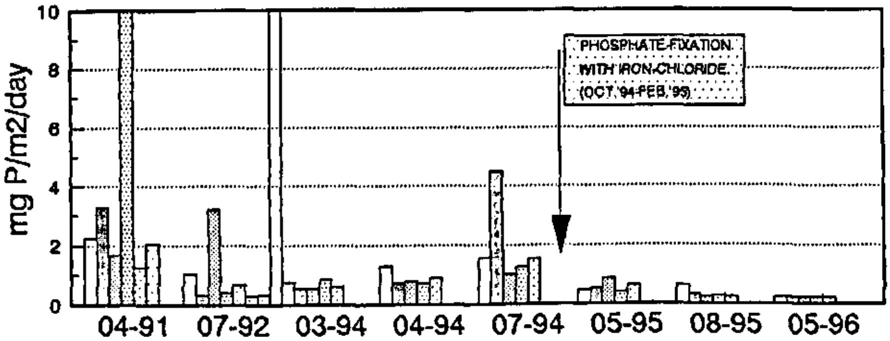


Fig. 4. Phosphorus realease before and after the measures.

Biomanipulation

The marking method seems to be a good method to determine the fish standing stock and remvoval. This method can not be applied to small fish. The assumption about the domination by bream appaered to be true (Fig. 5). Local fisherman reported a more varied population after the biomanipulation.

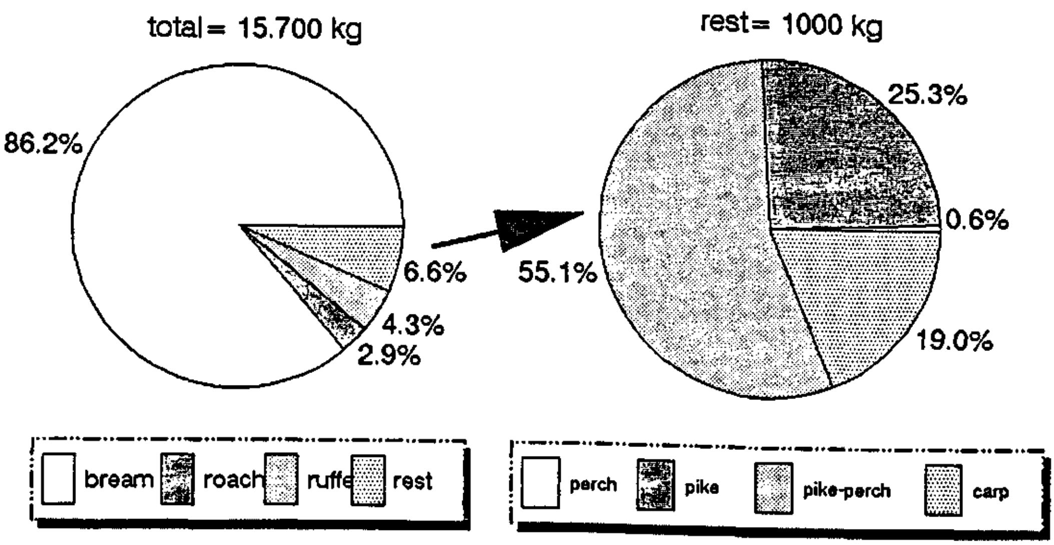


Fig. 5. Relative en absolute amounts of fish caught during biomanipulation.

Water quality

From the course of the mean summer values of some water quality parameters (Fig. 6) an improvement of the water quality was observed. Depth of Secchi-disc transparency doubled and total phosphate-, total nitrogen- and chlorophyll-concentrations decreased to 40%, 60% and 50%, respectively, of the values before restoration. None of the values fit the specific ecological standards and the standards for swimming water (transparency of 1 m.) . Soon after the biomanipulation there was a transparency of about 1 m in the whole lake during two month's. After the first improvement of water quality since measures were taken, a slight deterioration occurred in 1996.

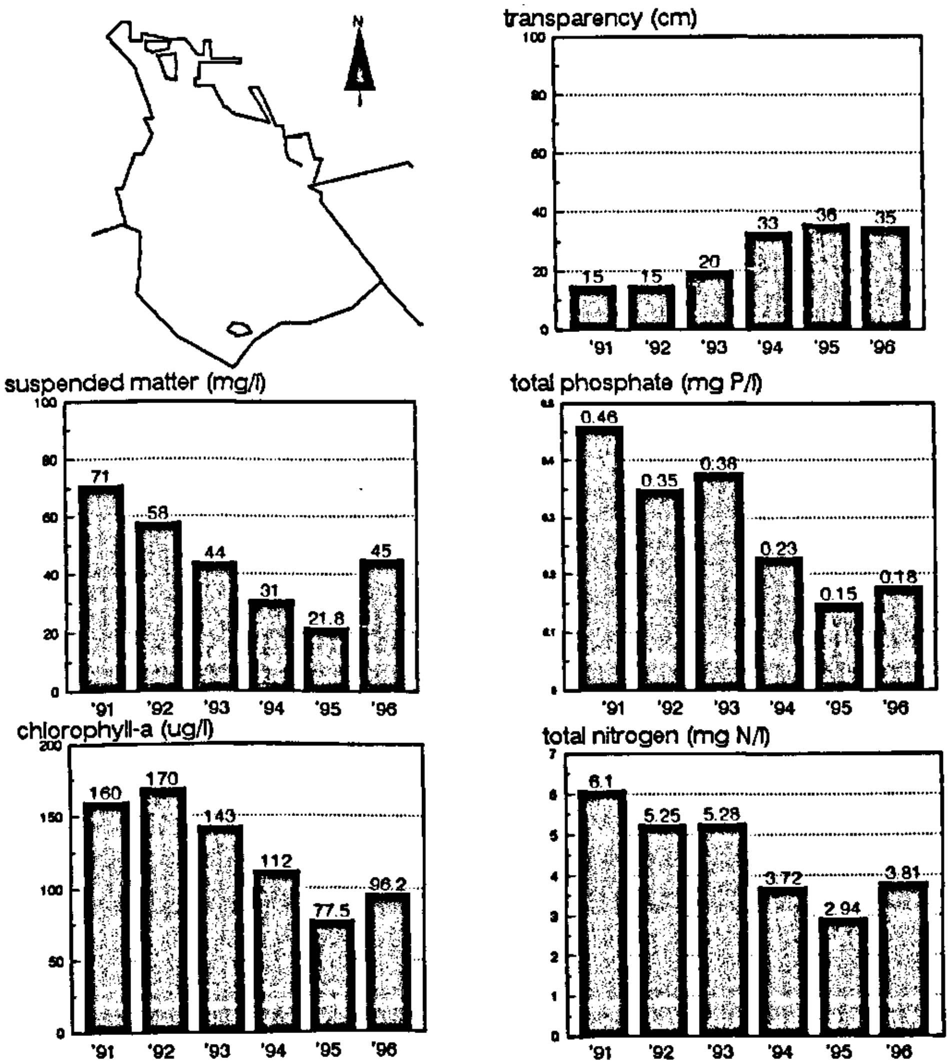


Fig. 6. Water quality parameters of the lake in the period 1991-1996.

The effect of restoration on the algal biomass and total-phosphorus concentrations in the lake is also clear from the plots of chlorophyll-a against total-phosphorus in different successive periods (Fig.7.). The decrease of both chlorophyll-a and total-phosphorus since 1981 is very distinct. Up to 1993 there is no correlation between chlorophyll-a and total phosphorus concentrations. After the restoration the algal biomass seems to be controlled by the total phosphorus concentration.

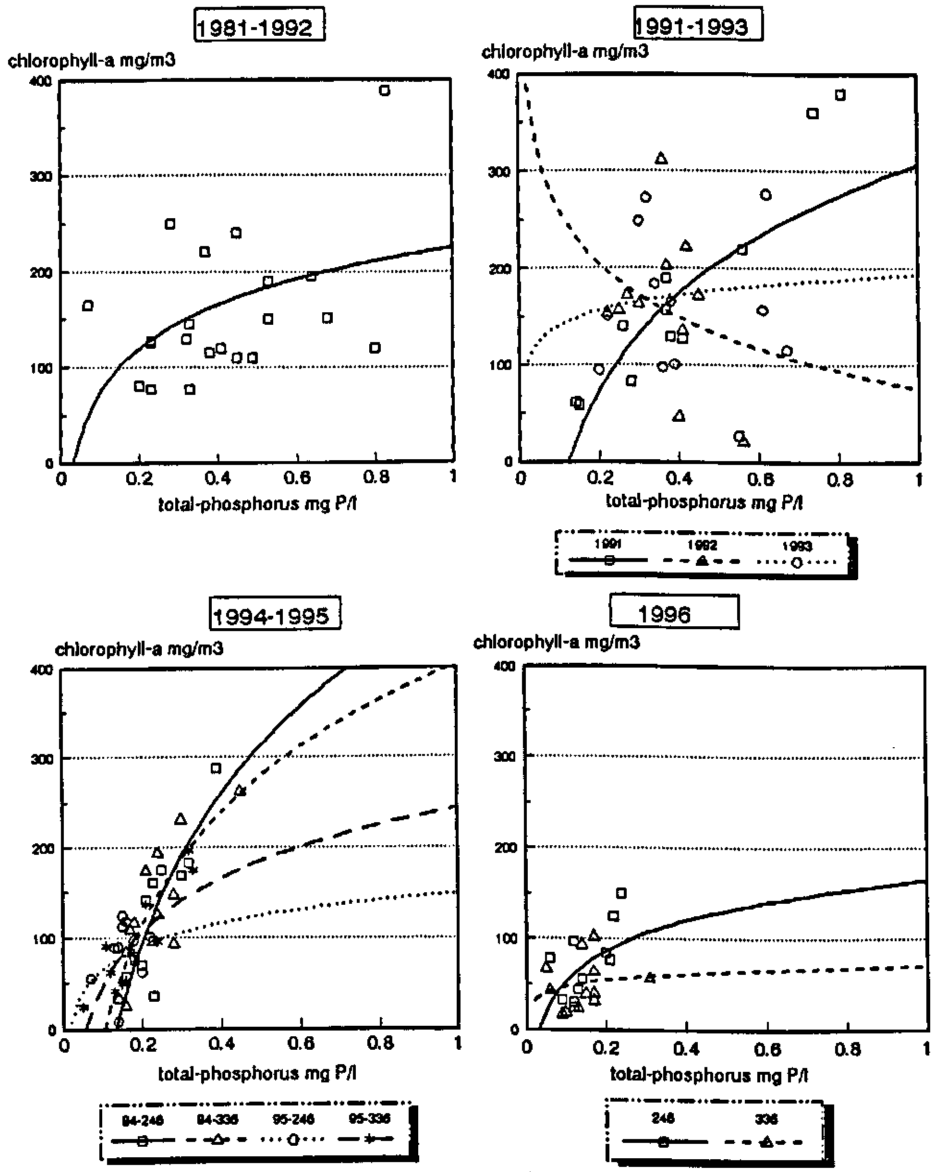


Fig.7. Chlorophyll-a concentrations plotted against total-phosphorus concentrations in the lake from 1981 up to 1996. (Logaritmic regression lines are indicated).

CONCLUSIONS

The hydrological isolation is successful. The advantage is that this facilitates supply of purified water. A disadvantage is the increase in the residence time of the water in the lake. This aspect might play a negative role for the development of the water quality.
The reed filter works as filter for suspended solids and to lesser extent for filtering nutrients.
There are still some problems with the chemical treatment. It is suggested that in this case phosphate cannot react with iron chloride because it is complexed by the humic compounds in the water.
Technically, the dredging was the most difficult measure to control. From measurements on the phosphate release we can conclude that phosphate fixation contributed to it.
Up to now (July 1997) we do not know about the development of the fish population since the biomanipulation in the early spring of 1995. It can be assumed that the clear period in 1995 is the effect of the fishing. In this clear period water submerged macrophytes were stimulated to grow. The absence of water plants is a possible reason for the slight deterioration of water quality in 1996.
In the future the fish population will be surveyed and the introduction of water plants is planned.

ACKNOWLEDGEMENT

Dr. Ramesh D. Gulati (Centre of Limnology, Netherlands Institute for Ecological Research, Nieuwersluis) is thanked for critical reading the manuscript and improving the English text.

EUTROPHICATION ABATEMENT IN LAKE NAARDERMEER

N.W. Broodbakker & P.T.J.C. van Rooy

DHV Water BV, P.O. Box 484, 3800 AL Amersfoort, The Netherlands

ABSTRACT

Lake Naardermeer is a nature reserve surrounded by farmland, roads, railways and towns. Thus, many actors are involved in possible restoration measures and agreement had to be reached between these actors about which measures need to be taken and how, when, and by whom these measures should be executed and paid for.
A comprehensive plan for the ecological recovery of Lake Naardermeer has been prepared and a covenant between the many actors involved has been signed. The covenant establishes the selection of measures which need to be taken, divides responsibilities and ensures sufficient financial means and cooperation for the execution of the selected measures.
The main bottle necks and their probable causes were identified. Possible measures were worked out and thoroughly studied to judge their effectiveness in solving the bottle necks as well as their technical, administrative and financial feasibility. This was followed by a lengthy process of study, consultation of experts, deliberation, and reiteration in order to select the most feasible recovery measures and to establish a Recovery Plan which is supported by all organisations and administrations involved.

KEYWORDS

Covenant, dessication, eutrophication, Lake Naardermeer, wetland restoration, water management.

INTRODUCTION

Lake Naardermeer is a complex of small lakes, swamp forest and reedlands of about 700 ha in the centre of the Netherlands. It is the northernmost of a chain of shallow lakes and marshlands east of the river Vecht in the southeast of the province of North-Holland (fig. 1). This area of lakes and marshes is situated on the borderline of a higher sandy ice-pushed ridge "The Gooi" and the low lying peaty "polders" along the river. It is an internationally important wetland nature reserve in terms of the RAMSAR-convention and one of the most important freshwater wetlands in the Netherlands, both botanically and because of its rich bird communities. Lake Naardermeer has a natural origin unlike most of the other lakes in the area which were formed by peat mining/cutting. In the middle ages it used to be in open connection with the Zuyder Sea via the river Vecht.

There have been several attempts to drain the lake in the 17th and at the end of the 19th century, but these were eventually abandoned because of intensive seepage of groundwater and therefore high costs of drainage. In the end Naardermeer was left to itself and again became a wetland area of open lakes and swamps alternating with reed- and hayland with a wide variety of plant life providing a rich habitat for many types of wildlife and especially water, swamp and reedland birds.

In 1904 the council of Amsterdam proposed to use Naardermeer as a refuse dump. These plans were abandoned and the protests led to the erection of the Society for the Protection of Nature with as its first purchase lake Naardermeer. It was subsequently managed as a nature reserve, the first of its kind in the Netherlands.

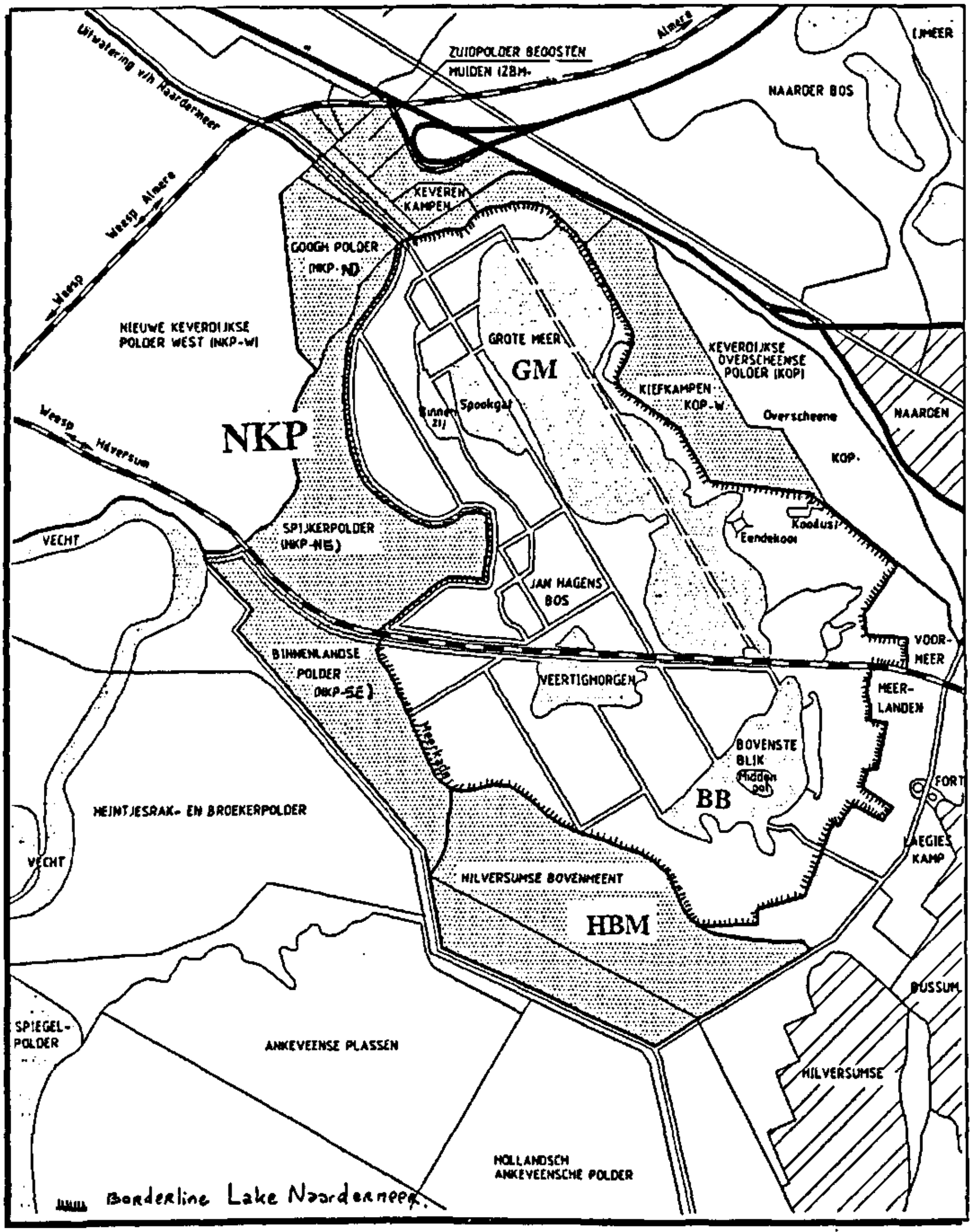


Fig. 1. Map of Lake Naardermeer and its surroundings

MAIN ISSUES

The sandy ice-pushed ridge of the Gooi provides the nutrient poor and bicarbonate rich seepage which is essential for the distinctive plant- and wildlife of Naardermeer and the other wetlands in the river Vecht area. The amount of seepage has decreased significantly as a result of increasing extraction of groundwater to provide drinking water and the decreasing amount of infiltration due to nearby urbanisation and the accompanying "hardening" of the soil.

On the other hand, the agricultural activities in the western polders and the draining of two former lakes have led to the subsequent semi-continuous lowering of groundwater levels, an effect increased by the settlement of the peaty soil and the reclamation of the large polder of Flevoland (Witmer, 1989).

Because of the porosity of the soil of lake Naardermeer this has led to increasing drainage to the surrounding polders, which together with the diminishing seepage, has led to severe water shortages in the dry summer period. To prevent drying out the lake had to be supplemented with water from the nearby river Vecht. In the sixties this water used to be rich in nutrients and micro-pollutants, which has led to eutrophication, degradation of plant life as well as water and sediment quality, a high incidence of FLAB (Floating Algal Beds) and recurrent blooms of blue-green algae. Water suppletion was stopped for a number of years for these reasons but had to be resumed because the resulting desiccation of the area was even worse (Schot et al., 1987).

In order to provide water of a higher quality, a dephosphatizing plant was build. This has resulted in a much better water quality in the northern part of the lake, a partial improvement of water quality in general and a partial recovery of wetland and water vegetation. These effects were not however strong enough to ensure the long term recovery of the lake.

Therefore the Province of North-Holland, the regional water boards, the councils of the towns surrounding Lake Naardermeer and the Society for the Preservation of Nature, the owner of the lake, decided to combat together the further decline of lake Naardermeer. This has lead to an integrated approach to the eutrophication abatement in Lake Naardermeer resulting in a recovery plan to restore the nature values of Lake Naardermeer.

BOTTLE NECKS

A comprehensive approach in eutrophication management means that many aspects of the water system, its functions and social as well as economical aspects have to be taken into account (van Rooy, 1996). This means that attention must pe paid to ground water as well as surface water and to quality as well as quantity, while also taking note of the different functions and uses of ground and surface water in and around Lake Naardermeer and of developments in the surrounding towns and infrastructure. Therefore requirements in the policy fields of water, environment, and physical planning had to be addressed and the amount of public and administrative support for possible measures and the economic feasibility of executing the measures had to be examined. The coherent study of all these aspects has lead to an administratively and publicly widely supported Recovery Plan involving many different aspects and actors (Broodbakker, 1993). The main focus points which had to be addressed in the study are indicated in the following.

Water quantity
- decrease of seepage because of groundwater extraction and hardening of surfaces (reduced recharge) in the seepage supplying sand ridge "het Gooi";
- increase of infiltration to surrounding polders because of low water tables;
- net seepage has changed into net infiltration;
- water shortages in the summer period which have to be supplemented with surface water rich in ions and nutrients (Schot, 1988).

Water quality
- water suppletion with ion and nutrient-rich water;
- internal loading with phosphorus and nitrogen caused by desorption from the sediment;
- seepage of brackish water leading to a higher chloride content in the southern part;
- decomposition of peat caused by desiccation, leading to higher nutrient contents of the surface water;
- nitrogen pollution of groundwater in the seepage supplying sand-ridge (recharge area) "het Gooi";
- acidification and enrichment with nutrients caused by atmospheric deposition (Schot et al., 1987).

Sediment
- substantial amounts of sludge in most parts which are stirred up by wind, leading to a further deterioration of water quality;
- in several places the sediment had a high content of nutrients, DDT and/or heavy metals.

Ecology
- reduced diversity and cover of water and wetland vegetation, especially seepage-dependent vegetation;
- increasing cover of swamp forest (elder, willow) at the expense of other types of wetland vegetation (partly caused by desiccation and natural progression);
- increasing incidence of FLAB and recurrent bloom of blue-green algae;
- dying-off of Characea and other water plants, caused by reduced transparency of the water;
- damage to water and seepage dependent vegetation because of high ion-content of the water (especially

chloride and sulphate) (Wassen et al., 1989; Barendregt et al., 1995);

Physical planning & infrastructure
- increased urbanisation;
- construction and extension of roads and railways;
- disturbance of fauna by sound-pollution.

Policy
- lack of legal instruments to execute many recovery measures within a limited period of time;
- accelerating the purchase of bordering agricultural land (in order to raise water levels);
- limited social support to raise water levels in surrounding areas.

INITIAL RECOVERY MEASURES

A number of measures had already been commenced or executed before the Recovery Plan was initiated, the main ones being:
- surplus drainage in the surrounding polders was reduced;
- hydrological isolation of the very large cormorant colony (more than 5000 birds), to prevent further eutrophication;
- pumping back polluted superficial seepage water from a neighbouring canal;
- dephosphatizing suppletion water;
- dredging of one of the southern lakes;
- governmental allotment/allocation of most surrounding areas as "Reserve Area", which means they can be bought with governmental funds, but only if the owner volunteers to sell, and handed over to the Society for the Preservation of Nature;
- purchase of/acquiring agricultural land in these "Reserve Areas" to serve as a future water buffer area by increasing the water level after purchase of most lots in each polder;
- temporarily damming off ditches in order to maintain higher water levels in the acquired parts of these "Reserve Areas";
- provincial measures to reduce the amount of groundwater withdrawal in the sand-ridge area "het Gooi".

However, further measures are necessary to ensure that Lake Naardermeer will remain a valuable and species-rich wetland of international importance.

RESULTS & DISCUSSION - RECOVERY PLAN

A number of steps were required to work out a feasible Recovery Plan. The objectives and current situation had to be established. The main bottle necks had to be identified by comparing the current situation with the objectives. This is illustrated in figure 2 and table 2 for a number of selected parameters. Secondly, the probable causes of the bottle necks had to be established and possible solutions had to be worked out. The resulting set of measures was thoroughly studied and judged to ascertain their potential effectiveness towards solving the bottle necks and their technical, administrative and financial feasibility. A lengthy process of study, consultation of experts, deliberation, and reiteration was necessary to select the most feasible recovery measures and establish a Recovery Plan which was supported by all organisations and administrations involved in the process.

Perhaps the most important part of this process was the consultation and coordination of all organisations and administrations involved. This was carried out by setting up meetings in which the proposed measures were discussed, deliberated, and judged, involving several levels of administration within each organisation. The basic discussions and deliberations took place in a so called "working group" consisting of technically knowledgeable employees of the organisations involved. The next phase was deliberation on a higher level of administration within the organisations, and the final phase consisted of deliberations on the highest level of administration.

The resulting set of measures and the criteria on which they were judged are presented in table 1. The main selection criteria were: speed of realisation or time to effect, effectiveness, total environmental yield, and technical, financial and administrative feasibility and desirability. In the following a short description is given of the main measures of the Recovery Plan. Most of these measures have recently been executed or are being executed. Others are part of an process of execution which will continue into the next decade. The corresponding numbers of the measures of table 1 are indicated in the text in parentheses.

Table 1. Criteria used for testing cost-effectiveness and feasibility of possible recovery-measures in and around Lake Naardermeer

	MEASURES	TIME TO EFFECT			EFFECTIVENESS				FEASIBILITY			COST	EXE-CUTE
		before 1995	1995 to 2000	after 2000	Water quality	Water quantity*	Fauna	Flora	Tech-nical	Financial	Admini-strative		
1	Reducing groundwater extraction in Het Gooi		x		+	125-275	+	++	++	o	+	++	yes
2A	Construction of separate sewer systems in Het Gooi			x	o	50-100	+	+	-	-	-	+	no
2B	Increasing infiltration of rainwater in Het Gooi		x		o	+	+	+	++	-	+/-	+/-	no
3A	Increasing water level in Nieuwe Keverdijkse Polder-East (NKP-E)		x	x	o	100-500	+	++	++	+	+	++	yes
3B	Increasing water level in Hilversumse Bovenmeent (HBM)		x	x	o	100-500	+	++	++	+	+	++	yes
3C-D	Increasing water level in KOP-W en ZBM-SE			x	o	50-150	+	++	++	+	+	+	yes
3E	Ending extra pumping in NKP-W			x	o	+	o	+	++	-	-	+	no
3F	Increasing water level in ditches along the dikes of Naardermeer	x			o	+	o	++	++	+	+	++	yes
4	Varying the water level in Lake Naardermeer	x			-	+	+	-/+	++	++	+	-/+	no
5	Improving the dikes around Naardermeer		x		o	o/+	o	o	++	-	+	o	no
6	Alternative sources and locations for suppletion		x		o/-	o	o	+	+	-	+	-/+	no
7	Sanitize and assign Soil Protection Areas in the areas around Naardermeer	x	x		++	o	o	+	++	-	+	++	yes
8	Damming off ditches in Meerlanden en Voormeer	x			+	-	o	+	++	+	+	+	yes
9	Binding phosphorus in the sediment	x			+	o	+	+	++	+	+	+	no
10	Dredging sludge	x	x		++	o	++	++	++	-	+	++	yes
11	Catching white fish	x			o/+	o	o/+	o/+	++	+	o	o/+	no
12	Removing large pike and bream	x			+	o	+	+	++	+	o	+	yes
13	Adapting the reed cutting regime	x	x	x	o	o	++	++	-/+	-/+	o	++	yes
14	Restoration by digging out swamp-forest and reedlands	x	x	x	o	o	++	++	++	+	o	++	yes
15	Acellerating aquisition of Reservation Areas	x	x	x	o	++	+	++	++	+	-	++	yes
16	Assign Reservation Areas	x			o	++	+	++	++	o	++	+	yes
17	Well considered physical planning of townships and infrastructure	x	x	x	o	o/+	++	+	++	o	-/+	+	yes
18	Improving ecological infrastructure in surrounding areas	x	x	x	o	o	++	+	++	-	+	++	yes

Judgement: ++: very positive; +: positive; o: no effect; -: negative; --: very negative. The full names of the abbreviations used can be found in fig. 1.
*) Expected decrease in suppletion in 1000 m^3 each summer half year, caused by increasing seepage or decreasing infiltration (+ = positive but can not be quantified.

Reducing groundwater extraction in the seepage providing sand-ridge "het Gooi" (1). Groundwater extraction to provide drinking water will be halved before 1998 and industrial use of groundwater will be strongly reduced. This will result in the restoration of the groundwater flow to and seepage in Lake Naardermeer and the other wetlands of the river Vecht area. It will also result in a reduction of desiccation, the seepage of brackish water, the need for water suppletion, and in a recovery of seepage related vegetation (Witmer, 1989).

Stimulating the infiltration of rainwater in "het Gooi" (2). The construction of improved separate sewer systems when building new town districts in combination with infiltration of the 'clean' rainwater fraction by way of infiltration basins or wetland systems. Where possible the infiltration of clean rain- and surface water will be stimulated by the surrounding town councils. These measures will increase the amount of groundwater in the recharge area and therefore the seepage in lake Naardermeer and the other wetland areas in the seepage zone.

Assigning and acquiring of Reserve Areas (15-16). The largest part of the area surrounding Naardermeer has been designated as 'Reserve area'. Being designated a 'Reserve area' makes it possible to buy the land using government funds, providing the owner agrees. This is necessary to provide a more effective buffer area around the lake to maintain the desired water level in Lake Naardermeer by increasing water levels in the surrounding polders. It also prevents future surface or groundwater contamination risks from activities in the areas directly surrounding the lake. Large parts of the Reserve Areas were still privately owned and had to be acquired offering financial compensation or land in exchange. Acquisition of the remaining privately owned land is a prerogative to increase water levels in the polders surrounding Naardermeer.

Assign, clean up and protect Soil Protection Areas (7). The groundwater in the infiltration area west of Lake Naardermeer provides the seepage in the Lake. It is therefore important that this area is protected against groundwater pollution. It is also important that existing polluted areas are cleaned up. This will be achieved by assigning them the status of Soil Protection Areas and by taking further soil protection measures. These measures will preventing a further deterioration of seepage water quality.

Increasing water levels in the polders surrounding Lake Naardermeer (3). The water levels in the acquired terrain in the Reserve Areas has been increased and will be increased further after acquisition of all the surrounding polders has been completed. This will reduce the loss of water through infiltration from Lake Naardermeer to the surrounding polders. It will improve the water balance and water quality of Lake Naardermeer, and prevent further desiccation and eutrophication of the lake (Witmer, 1989).

Damming off ditches southeast of Naardermeer (8). Some ditches in the polders southeast of Naardermeer were directly connected with the lake. The water in these ditches is polluted with nutrients and other pollutants. The damming of these ditches has prevented further eutrophication of Naardermeer.

Sludge and nutrient-removal by dredging (10). The top-layer of sludge in most of the lakes has by now been dredged to prevent amelioration of surface water quality by the release of nutrients from the sediments. This release is stimulated by wind, activities of fish prodding in the soil and by anaerobic conditions. The dredging prevents further pollution and enrichment with nutrients of the surface water in Naardermeer. It also increases water depth, lessening the resuspension of sediment and provides better growth conditions for higher plants and Characea, and better living conditions for Otter and Pike.

Designing and constructing ecological corridors (18). Ecological corridors will be constructed to connect Lake Naardermeer with the other wetland areas and lakes in the Vecht-area in the south-west, and with Lake IJmeer in the north, making possible safe migration of animals (and plants) between surrounding wetland and lake areas and Lake Naardermeer.

Preventing infrastructural developments around Naardermeer (17). Amelioration of the condition of Naardermeer by construction or extension of roads, railways, and other infrastructural developments will be prevented or minimalised by taking compensating measures. The main effects will be prevention of further faunal disturbance, pollution and desiccation, as well as the prevention of breaking up ecologically important areas and corridors.

Phosphate-removal in suppletion water. Phosphate-removal of suppletion water will continue as longs as water suppletion of Lake Naardermeer remains necessary. This measure has lead and still leads to a continuous improvement of the surface water quality of Lake Naardermeer, the prevention of algal blooms and the recovery of water and wetland vegetation.

Monitoring plan. Setting-up and executing a monitoring plan to monitor the changes in water quantity and quality and ecological changes in fauna and flora. The results of the monitoring should be compared with the target images for a selected number of parameters. The main parameters and their present deviations of the target values are qualitatively indicated in fig. 2.

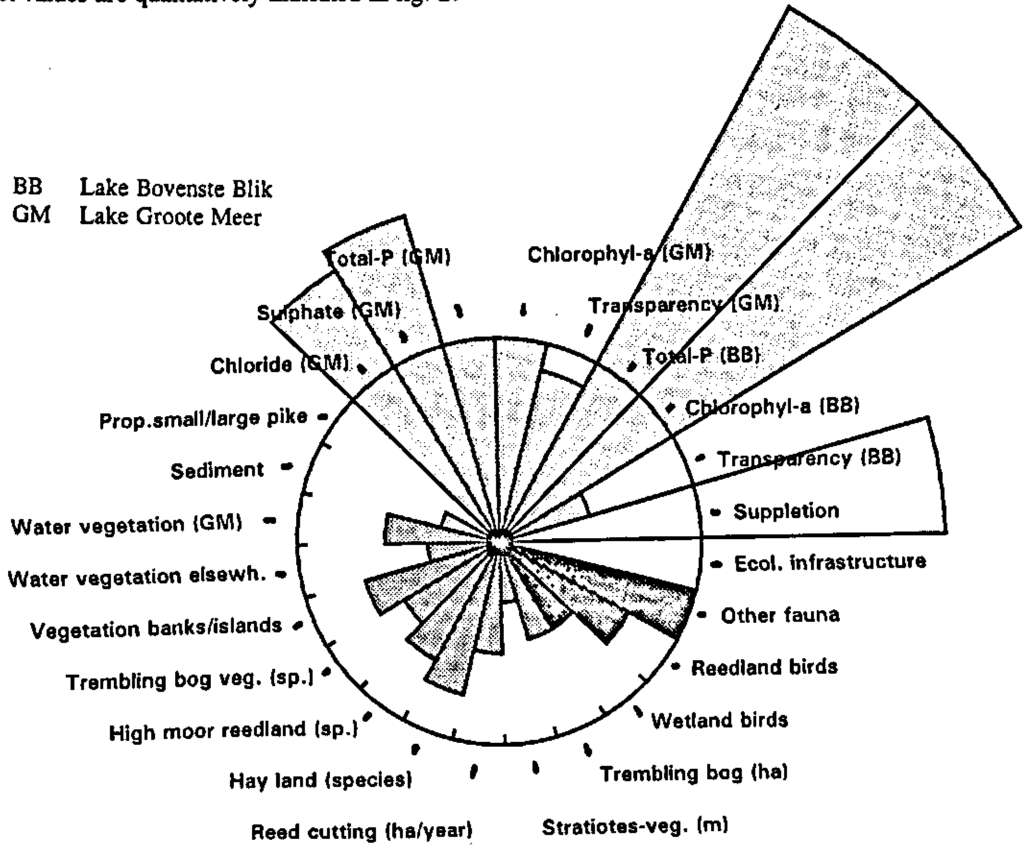


Fig. 2. AMOEBA - Lake Naardermeer - target image and objectives

AMOEBA : A general Method Of Ecological and Biological Assessment.
The edge of the circle presents the target image (value of objectives: 100%) in the year 2005. The length of the 'pie'-wedges indicates the deviation of the target image (values). The current situation can deviate in two directions. Either more or less than 100% of the target value. For example the current concentration of total-phosphorus (total-P) is much higher than the objective, while current transparency is much lower than desired, especially in Lake Bovenste Blik (BB).

Financial and execution responsibilities. Another important part of the Recovery Plan was to divide responsibilities with regard to the execution and the division of the financial costs of each of the measures. In most cases more than one party is responsible for the execution and the financial support of each respective measure and executing and paying parties are not always necessarily the same.

Communication Action Plan. As a last but very important final step a Communication Action Plan (CAP) was worked out to involve and inform the local and larger communities and organisations with interests in the area as well as the general public. This includes a PR (Public Relations) campaign, coordinating communication, meetings, articles for journals and other activities.

CONCLUSIONS - EXECUTION OF THE RECOVERY PLAN

Execution of the Recovery Plan will only be possible if all organisations involved agree on one communal view. This includes an integrated, shared vision, and agreement on problems and possible solutions. Furthermore a coordinated effort is needed to use legal instruments and to provide the financial means needed to execute the agreed recovery measures. This concerted view was presented in a so-called "Executive" or

"Nature Covenant" in which all organisations/parties involved have committed themselves by signing an agreement to execute the Recovery Plan and to provide the financial means and organisational back-up necessary to execute the chosen measures. This agreement was signed in 1993 and the execution of the Recovery Plan is in full action at the present time.

When dealing with such a complicated Recovery Plan it is important that one party is responsible for the coordination of all efforts, a task adopted by the Province of North-Holland. A Communication Action Plan was developed as part of the Recovery Plan to ensure wide public and administrative support, especially in the district and surrounding towns.

An organisation named "Cooperative Naardermeer" was established to guarantee the effective future coordination and execution of the Recovery Plan. All parties involved are represented in this organisation. The group meets once or twice each year or when one of the parties involved feels a meeting is necessary to decide about some urgent or important issues. The Province of North Holland is responsible for the execution of the Communication Action Plan as representative of the Cooperation Naardermeer.

The final result of this concerted effort has been the continuing recovery of one of the most important wetland areas in the Netherlands. It is an effort not only resulting in the recovery of plant and bird life, but it also guarantees the continuous enjoyment of this jewel in the crown of the Dutch wetlands by the inhabitants of our small and densely populated country.

ACKNOWLEDGEMENTS

The Province of North Holland, the Waterboard Amstel, Gooi & Vecht and the Society for the Preservation of Nature in the Netherlands, initiated the project together and financed the preparation of the Recovery Plan.
The Recovery Plan Naardermeer could not have been prepared and written without the concerted effort of numerous representatives of authorities and organisations involved in the management of Lake Naardermeer and its surroundings. They supported and supervised the preparation of the plan as members of a "Working Group" a "Project Group" and a "Steering Committee".
Besides the already mentioned authorities and organisations the following were also involved in preparation of the Recovery Plan and are involved in its execution. They are: the city councils of Bussum, Hilversum, Muiden, Naarden and Weesp, the Regional Council for the Gooi and Vecht Areas, Dutch Railways, the DLG (Service Rural Areas), the Agricultural Board, the Ministry of Agriculture, Nature Management and Fisheries and the Ministry of Public Works and Water Management (Rijkswaterstaat).

REFERENCES

Barendregt, A., M.J. Wassen & P.P. Schot (1995). Conservation of wetlands depends on hydrology, illustrated in the nature reserve Naardermeer. *Biological Conservation*, 72, 393-405.

Broodbakker, N.W. (1993). Recovery Plan Naardermeer . DHV Water BV, Amersfoort (in Dutch).

Rooy, P.T.J.C. van (1996). Towards comprehensive water management in The Netherlands (3) Planning. *European Water Pollution Control*, 6 (3), 36-45.

Schot, P.P. (1988). Groundwater system analysis as a basis for wetland management in the central Netherlands - case: the wetland Naardermeer. In: Ruzicka, M., T. Hrnciarova & M. Miklos (eds.). *Proc. VIIIth Int. Symp. Problems of Landscape Ecological Research*, Bratislava, 1, 187-197.

Schot, P.P., A. Barendregt & M.J. Wassen (1987). Hydrology of the wetland Naardermeer: influence of the surrounding area and the impact on vegetation. In: J.W. van Hoorn (ed.), Agrohydrology - recent developments, Elsevier, Amsterdam.

Wassen, M.J., A. Barendregt, M.C. Bootsma & P.P. Schot (1989). Groundwater chemistry and vegetation of gradients from rich fen to poor fen in the Naardermeer (the Netherlands). *Vegetatio*, 79, 117-132.

Witmer, M.C.H. (1989). Integral water management at regional level. An environmental study of the Gooi and the Vechtstreek. Ph.D. Thesis University of Utrecht, 's-Gravenhage.

RESULTS OF LAKE RESTORATION IN THE GEERPLAS

E.H.S. van Duin, L.J. Frinking, F.H. van Schaik & P.C.M. Boers

Water board of Rijnland, P.O. Box 156, 2300 AD Leiden, The Netherlands
Institute for Inland Water Management and Waste Water Treatment, P.O. Box 17, 8200 AA Lelystad, The Netherlands

ABSTRACT

In the Langeraar lakes, three small connected lakes in the western part of the Netherlands, lake restoration was started in 1986. Both the internal and external phosphorus load to one of the lakes, lake Geerplas, was reduced. During 6 years after the measures were taken, the water quality in the Geerplas and in one of the other lakes has been monitored. Water quality improved in the Geerplas in the sense the phosphorus concentrations and turbidity decreased during 3 years. In the end of the third summer, phosphorus concentrations increased rapidly, presumably due to excessive phosphorus release from the lake sediment. High phosphorus concentrations remained for almost two years, than decreased slowly. Phytoplankton biomass started to descend during the same year and phytoplankton composition changed as well.. Zooplankton biomass varied during the years. After two years of high phosphorus concentrations, concentrations dropped without any additional measures taken. Chlorophyll-a concentrations and turbidity are at the same level now as shortly after the measures were taken. Lately, the effects of project measures are evaluated and additional measures are considered

KEYWORDS

Cyanobacteria, eutrophication, lake restoration, phosphorus, phytoplankton, zooplankton

INTRODUCTION

In the low-lying centre of the Netherlands, several broad areas are located. These shallow water systems are valuable nature conservation areas. Most of them are severely affected by eutrophication, resulting in high biomass of phytoplankton (cyanobacteria), turbid water and almost no macrophyte growth. To restore the natural values of these areas, including the water quality, many lake restoration projects were started in the period 1985-1995 (van der Does et al, 1992, Boers et al, 199., Hosper, 1997). In these projects various combinations of measures were implemented to reduce internal and external phosphorus loading were implemented. Evaluation of the projects may lead to a reconsideration of the expected success rate of certain measures and the need for additional measures. In this paper the evaluation of a lake restoration project conducted for a small lake is described. Measures, water quality results and future strategies are included.

DESCRIPTION OF THE LAKE RESTORATION PROJECT

The Langeraar lakes are three small connected fen lakes in the Western part of the Netherlands. The area of the Geerplas is the most northern part, with a surface area of almost 30 hectares, consisting of about 24 hectares of open water and 6 hectares of grass land. The average depth of the lake is 1.9 metre and variations in depth are minimal, (lake volume above $600m^3$). The lake sediment consists of mainly of peat and some clay. Until 1990, the Geerplas and the central lake, named Noordplas, were connected trough a narrow channel and water quality in all three lakes was similar (Frinking, 1992). After measures were taken lake Noordplas has been used as a reference lake for the Geerplas.

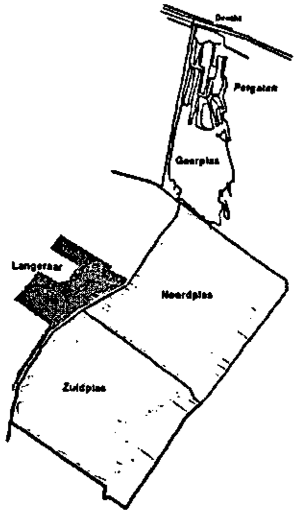


Fig. 1. The Langeraar lakes

Before the measures were taken, Lake Geerplas was characterised as a lake with a high external phosphorus load (< 0.9 $gP/m^2/y$) and a high net internal phosphorus load (2.0 $gP/m^2/y$). This high internal load was caused by a high phosphorus content of the top sediment (1.48 mg P/g sediment) and a low iron content of the lake sediment. Measures were taken in 1989 and 1990 and aimed at the reduction of both the internal and external nutrient load of the lake

- hydrological isolation of the Geerplas. This meant the construction of a flexible water barrier in the connection channel between the Noordplas and the Geerplas. The water level is maintained on a constant level, hence excess rainwater in winter has to be discharge and shortages in summer have to be replenished. Winter excess is pumped out of the lake into the river Drecht;
- the summer inlet water from the Drecht is dephosphorized. Dephosphorization is done by chemical treatment with Iron (III)-chloride. After sedimentation of Iron-phosphate-complexes, water is passed through a biofilter with reed. The average efficiency of the chemical treatment is 60% for total phosphorus and 10% for total nitrogen. Unexpectedly, the biofilter does not remove phosphorus but releases phosphorus to the passing water;
- reduction of all waste water discharges;
- isolation of the bird colony of blue herons and cormorants that is based in the Geerplas area;
- removal of the nutrient rich top layer of the sediment by dredging. The dredging of the fluid mud layer proved to be difficult and several techniques were used (Van der Does et al, 1992). A dredging technique in which screens were used to reduce the horizontal mud flow from undredged to dredged areas, proved successful in the removal of the mud layer.

DATA COLLECTION

The effects of the conducted measures are followed by an extensive monitoring program. Every four weeks the Geerplas and Noordplas were visited. During these visits water samples were taken at several sites and physical measurements were conducted. Samples were analysed for nutrients and heavy metals content. Some of the gathered data are presented in the figures below (figure 2, 3 and 4). Part of the obtained data were used to estimate mass balances for water and total phosphorus (Van Schaik et al, 1997). In figure 5 the monthly mass balance for the period after measures is presented. Reliable mass balances of the period for the measures are not available. Phosphorus release rates from intact sediment cores were measured according to a continuous flow technique (Boers & van Hese, 1988). For a period of

10 years one or two times each years, cores were sampled at three sites in the lake, results are presented in figure 9.
Counted data of phytoplankton species composition are available from 1987 till 1996 and presented in figures 6 and 7 (Van den Hove, 1997). The counted individuals are divided in the four main groups of phytoplankton, cyanophyta (blue-green algae), chlorophyta (green algae), chrysophyta (diatoms and chrysophytes) and other algae. The data from 1996 and 1997 are obtained from rough inspections of non conserved water (fresh material). Bioassays with the natural phytoplankton population are used to assess the growth limiting nutrient factor of the phytoplankton in Lake Geerplas and lake Noordplas.
In the data of the zooplankton species composition and biovolumes, a distinction is made between cladocera, copepoda and rotifera. Data are available from 1988 till 1996 (Aquasense, 1997) and presented in figure 8. The data collection program and sampling methods are described by Van Schaik et al, (1997).

EFFECTS ON WATER QUALITY AND LOADS UNTILL 1994

By May 1991, all measures are taken. By comparison of water quality variables obtained in the period 1987-1989 and 1991-1994 the effect of the measures is evaluated. As in the summer of 1994 radical changes in water quality variables occurred, the period thereafter is described separately.

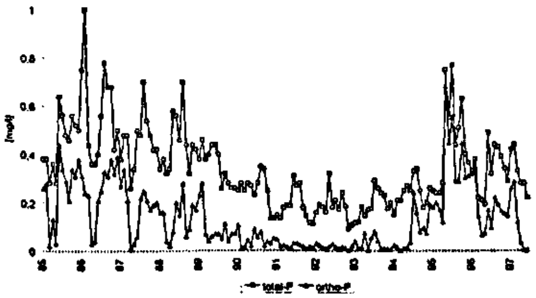


Fig.2. Phosphorus conc. in the Geerplas

The total phosphorus concentration in 1991-1993 varies around 0.2 mgP/l, which is significantly lower than the average value of 0.5 mgP/l, in 1987-1989, figure 2.Ortho-phosphorus concentrations approach zero and the phytoplankton growth seems to be limited by the availability of phosphorus.
The phytoplankton biomass in 1991-1993 reaches summer high values above 150 ug chlorophyll-a/l, (figure 3) but is significantly lower than previously to the measures. After the measures the variations in Secchi-disk depth are high, between 0.4 and 1.0 metre, figure 4. Before the measures were conducted, by values above o.4 metres were very rarely observed.

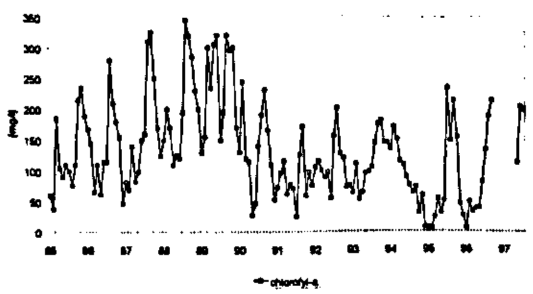


Fig. 3. Chlorophyll- a concentration in the Geerplas

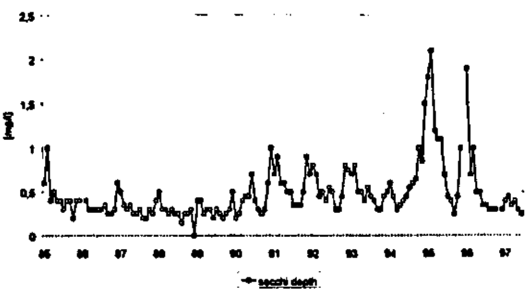


Fig. 4. Secchi-disk depth in the Geerplas

After the measures the water balance of the lake (table 1) is dominated by rainfall, evaporation and the remission of the rainfall surplus (pump). On a yearly base, a rain surplus occurs, however during the summer months an evaporation surplus occurs, which is replenished by the inlet of water from the river Drecht (inlet). The main source of phosphorus is still the inlet of water during the summer months, to supply the evaporation surplus. Remission of the rainfall surplus in winter, is the main factor to remove phosphorus from the lake. More phosphorus enters the lake than leaves it, hence a net accumulation of phosphorus to the lake occurs (residue).

Table 1. Average yearly water and phosphorus balance fro 1992-1995

IN	Water		Phosphorus		OUT	Water		Phosphorus	
	1000m3	%	kg	%		1000m3	%	kg	%
deposition	298	58	21	26	evaporation	234	46	0	0
inlet	164	32	39	48	pump	225	44	43	52
leaching	51	10	22	26	leaching	30	6	9	11
					infiltration	24	4	8	10
net release	0		0	0	storage	0	0	22	27
TOTAAL	513	100	82	100	TOTAAL	513	100	82	100

On a yearly scale no significant release of phosphorus occurs. However on a more detailed scale, figure 5 sediment water interactions dominate the phosphorus fluxes in the water system. If the net release is positive, phosphorus release from the sediment exceeds the phosphorus accumulation and concentrations in the water column increase (storage is negative).

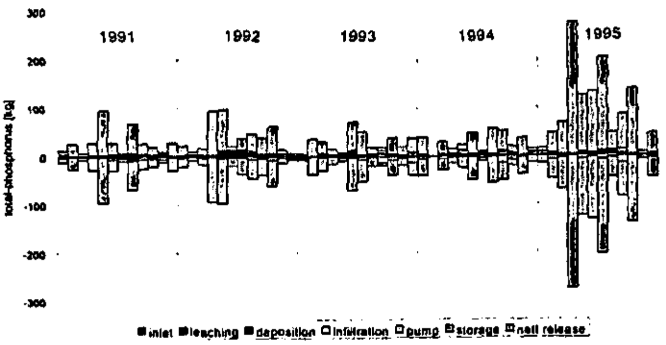


Fig. 5. Monthly phosphorus balance of the Geerplas, 1991-1995

The data of Lake Geerplas show a dominance of cyanobacteria from early spring throughout the summer (1988-1990), figure 6.

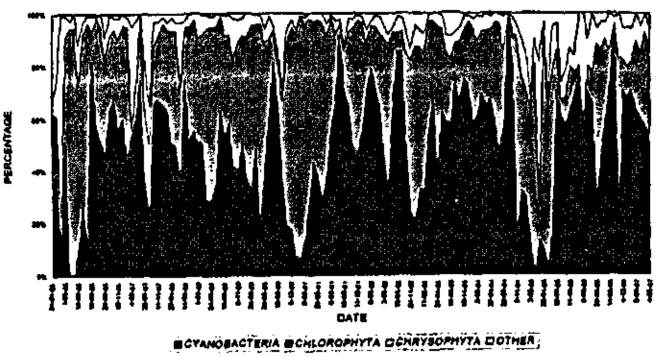


Fig.6 Distribution of phytoplankton groups in the Geerplas

The winter of 1991, autumn 1993 and a period from autumn 1994 till the summer of 1995 are dominated by green-algae. In spring 1987 and 1995 a dominance of chrysophytes, mainly diatoms, occurred. In the lake the cyanobacteria composition consists mainly of (> 40%) Lyngbya limnetica (figure 7). During the dredging activities in the winter 1990-1991 Lyngbya disappeared, but reappeared during the summer of 1995

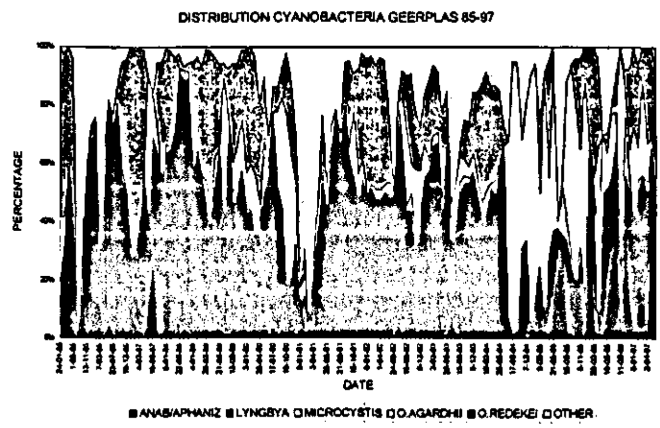


Fig. 7.Distribution of cyanobacteria in the Geerplas

Concentrations of crustaceae hardly change from 1988 till 1994. The rotifera concentration fluctuate during the years and reach high densities in 1990, 1992 and 1995 in both lake Geerplas and lake Noordplas. The grazing pressure from zooplankton on the phytoplankton community is not high, considering the concentrations and biovolumes of the zooplankton in comparison with other lakes in the Netherlands (Gulati, 1992). High grazing pressures are expected from large bodied zooplanktonspecies like the Daphnids. These species are hardly found in Lake Geerplas (and lake Noordplas).

WATER QUALITY EFFECTS AND LOADS AFTER 1994

In 1994 very spectacular changes in water quality in Lake Geerplas are observed. After a winter with a high phytoplankton biomass, of around 150 ug chlorophyll-a/l (figure 3), the biomass decreased gradually, until almost 0 ug chlorophyll-a/l in the winter of 1995. This biomass decrease was accompanied by a subsequent decrease in turbidity: the Secchi disk depth increased from 0.4 metres to lake depth (figure 4). Phytoplankton composition changed from a dominance of *Lyngbya* to a dominance of *Microcystis* during the spring of 1994 (figure 7).

One month before the phytoplankton dominance of *Lyngbya* was replaced, the zooplankton biomass increased from about 5000 um3/l to almost 50,000 um3/l. mainly caused by cladocerans (figure 8). These were *Diaphanosama brackyurum* and *Daphnia hyalina* complex. During the same months the ortho- phosphorus content in the lake increases, while the total phosphorus content remains at the same level of 0.2-0.3 mg/l. The exact sequence of the events is very interesting.

1. In the early spring, the chlorophyll-biomass and turbidity start to drop.
2. This is followed by a sudden increase in ortho-phosphate and zooplankton biomass is observed,
3. and subsequently a change in cyanobacteria -dominance.

It is not clear what event or process started the gradual decrease of *Lyngbya* biomass. Apparently this decrease of the phytoplankton biomass and the subsequent increase in clarity triggered a sequence of events that changed the ecology of the lake. Recently, suggestions are made that the collapse of the *Lyngbya* dominance might be caused by a virus infection. However this has not been explored yet.

The bioassay experiments show that the maximum yield decreased after the measures were taken. The phytoplankton population was N-limited before the measures were taken and both N and P-limited. June 1995, showed a very high yield of the biomass. At that time the population was probably light limited The available light energy and the relation with phytoplankton growth has not been a topic of study yet. Before a further management plan for the lake can be made, systematic analysis of the growth limiting factors will be needed

The phytoplankton composition is still slightly different from the phytoplankton composition before the 1994-1995 phenomenon. Although, the rough countings in 1996 and 1997 show that *Lyngbya limnetica* is always common, *Microcystis* is more often seen in the phytoplankton samples and *Lyngbya* is not so abundant any more. In autumn 1995 *O. agardhii* is the dominant cyanobacteria.

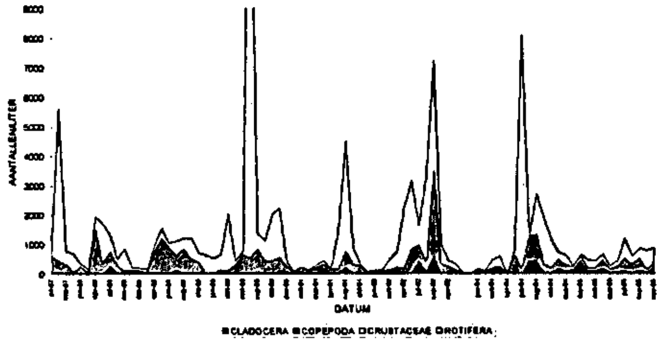


Fig. 8. Zooplankton distribution in the Geerplas

From April 1996 till now (June 1997) N-fixing algae are present (*Aphanizomenon* with heterocysts). The ratio's between the algae might shift when the samples are counted in our 'standard way. An exception are the winter months of 1994-1995. At that time relatively high amounts of Daphnia hyaline complex were found and very low Chlorophyll-a concentrations. The phytoplankton species composition consisted mainly of green algae (Van den Hove, 1997). These algae are considered to be a good food source for zooplankton (Gliwisz, 1990).

Phosphorus

The ortho-phosphorus content of the lake increased dramatically, up to 0.7 mgP/l in April 1995. As the measured phosphorus concentration remains high during the entire summer, an not a one sample result, this phenomenon should be taken seriously. An increase of 0.5 mgP/l in one month, is comparable with a flux of 300 kg. As no external factors that might effect the observed changes are known, in the phosphors balance of lake Geerplas the increase is contributed to a net release of phosphorus from the sediment. Under usual conditions non of the other sources in the range of this amount. The average inlet in 1992-1995 is 40 kg/y, deposition and leaching around 20 kg/y. Hence these sources can not account for an increase of 300 kg in one month.

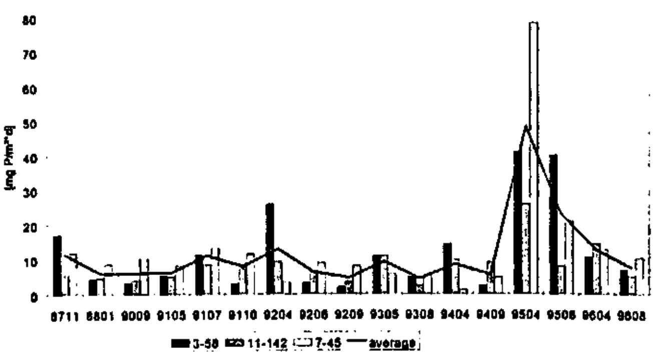


Fig. 9. Measured phosphorus release of the sediment

An increase of 0.5 mgP/l in one month, can be produced by a phosphorus release rate of 30 $mgP/m^2/d$ for 30 days. To assess the probability of the occurrence of such a high rate, this value was compared to measured release rates. The results are presented in figure 9. The release rate seemed not affected by the dredging experiment: From 1987 until 1995 lake averaged values varied around *XX* mgP/m2/d. Differences between laboratory values and mass balances estimated are due to several factors:

- mass balance estimates are net phosphorus release rates, where as measured values are gross rates. Hence laboratory results tend to be higher than mass balance estimates;
- laboratory experiments are conducted at a temperature range of 20-25 °C. In situ water temperatures will be much lower, especially in spring. At lower temperatures the release rates are lower. Again laboratory results tend to be higher than mass balance estimates;
- in continuous flow experiments, the water layer is diluted continuously. The measured release rate is therefor a maximum release rate;
- the laboratory experiments are conducted with three core samples, assuming these are a good representation of the lake sediment.

Summarising this, the conclusion can be made that in this case laboratory experiments and mass balance estimates will disagree, with probably higher values for the measured values. However, the observed substantial increase in both values for the April 1995 values is meaningful and suggests that such a phosphorus release from the sediment did actually happen.

Fish biomass

In 1989 and 1995 the fish stock was assessed in lake Geerplas. In both years the fish biomass was high, approximately 340 kg fish/ha, and consisted mainly of bream (70%). The dominance of bream is not surprising considering the eutrophication level of the lake. In the bream population fishes with a length of 10-35 cm were missing. This could be due to a weak recruitment of bream during the dredging activities or predation by the cormorants and blue herons living near lake Geerplas.

EVALUATION AND CONCLUSIONS

Project measures did not lead to expected water quality improvement. Although for several years phosphorus concentrations and phytoplankton biomass decreased, an ecosystem shift towards a more macrophyte dominated clear water state did not occur. Recently, total-phosphorus concentrations vary in a range of 0 and 0.8 mg/l. Phytoplankton biomass varies between0 and 250 ug/l chlorophyll-a and phytoplankton and zooplankton species composition are more dynamic. In the lakes there is hardly any macrophyte growth. Fish composition and biomass do not seem to be affected

.

The restoration project was based on the bottom up approach, with the effort directed to the reduction of phosphorus. The decrease in amount of inlet water in combination with the chemical treatment of the inlet water resulted in a substantial reduction of the external phosphorus load. However the efficiency of treatment is less than predicted, especially of the biofilter. To abstain from the inlet of water all together, and accepting the consequent water level variations might be considered.

The aim of the hydrological isolation of the lake and the treatment of the inlet water was to reduce the external nutrient loading. The internal load was not effectively reduced by dredging and with the increase in residence time, nutrient concentrations in the lake are completely dominated by the nutrient fluxes between sediment and water. The cause of these high phosphorus release rates, have to be further explored. Future measures to decrease nutrientconcentrations should either involve the sediment-water interactions or the residence time. For example, fixation of phosphorus in the sediment might be applied by the addition of iron. The residence time can be affected by extensive flushing of the lake. However, the current quality of the available water is not suitable.

The lake restoration was based on a reduction of phosphorus concentration, aiming at a phosphorus limited ecosystem. In the current state, periods of nitrogen and light limitation occur as well. A systematic analysis of the growth limiting conditions for phytoplankton over the years, including the phytoplankton response, is needed.

At this stage, the bottom up approach might be reconsidered. Additional top down measures can be considered, such as biomanipulation of the lake. Modification of the lake morphometry, by shore restoration or water level manipulation, might be examined as well.

References

AquaSense (1990-1997): Zooplankton in de Langeraarse plassen,. Rapport in opdracht van het hoogheemraadschap van Rijnland, rapportnummers 891043, 91.0129, 92.0214, 93.1317, 95.0716, 95.0716, 96.0960. *In Dutch*

Boers, P.C.M. & O. van Hese., 1988. Water Resouces-22: 355-363..

Boers, P.C.M., J. Van der Does, D.T. van der Molen & W.J. Rip, 1993. Eutrophication control in the Netherlands by reduction of P loadings: Approach, results and challenges for the future.
Bruning, C. and S.P. Klapwijk (1984): Application of ferivate spectroscopy in bioassays estimating algal available phosphate in lake sediments. Verh. Internat. Verein. Limnol. 22: 172-178.
Frinking, L.J. (1997). Vastlegging van de uitgangsituatie voor het baggeren van het noordelijk deel van de Langeraarse plassen: Geerplas. Intern rapport. HHR van Rijnland. *In Dutch*
Gliwisz, Z.M. (1990): Why do cladocerans fail to control algal blooms. Hydrobiologia 200/201: 83-97.
Gulati, R.D. (1990): Structural and grazing response of zooplankton community to biomanipulation of some Dutch water bodies. Hydrobiologia 2000/201:99-118.
Hosper, H. (1997): Clearing lakes an ecosystem approach to the restoration and management of shallow lakes in the Netherlands. Ph-D thesis Landbouw Universiteit Wageningen.
Van der Does, J., P. Verstraelen, P. Boers, J. van Roestel , R. Roijackers, 1992. Lake restoration with and without dredging of phosphorus enriched upper sedimetn layers. Hydrobiologia 233:197-210.
Van der Does, J. & L.J. Frinking, 1993. Effects of dredging of th phosphorus enriched upper sediment layers and reduction of the external loading in the Geerplas. 5th International Conference on the conservation and management of lakes. Stresa, Italy.
Van den Hove, L. (1997): Resultaten fytoplankton onderzoek 1987 tot en met 1995 in de Geerplas en de Noordplas. Intern rapport hoogheemraadschap van Rijnland.
Van Schaik, F.H., L.J. Frinking, E. de Groot & E:H.S. van Duin, 1997. Eutrofiëringsbestrijdingsproject de Geerplas- Evaluatie. Intern rapport HHR van Rijnland. *In Dutch*

NUTRIENT AND CADMIUM REMOVAL OF MUNICIPAL WASTEWATER BY WETLANDS

Ayçe Çelikel, Derya Boynukalın, Ülkü Kocaman, Yasemin Gün

Istanbul Technical University, Environmental Engineering Department, 80626 Maslak, Istanbul, Turkey

ABSTRACT

On June '96 the municipal wastewater has started to be treated by lab-scale constructed wetland and the studies have been lasted till October. In this period of time; nutrient, organics and cadmium removal efficiencies of the system have been monitored. After 1 month adaptation period, the time during June to October '96 ; TKN, TP, BOD_5, COD, TSS, Cd, concentrations at the inflow and outflow water have been found by Standard Methods. Obtained results indicate that in most of the experiments: Cd added to the wastewater is 0.5 mg/l and the removal efficiency is 95 %. Removal efficiency of COD is 75-80 % . Because of the feeding tank which is run as a presettling tank, 30 % of organics have been treated before discharging to the system . Some of the settleable organics is treated by anaerobic conditions and the rest of them is removed by aerobic process and infiltration. Removal efficiency of TP is 80 %. Nitrogen in the wastewater is in the form of NH_3-N and org-N but some of the effluent -N is found in the form of NO_3-N, the existence of nitrification & denitrification process. Also nitrogen removal occurs by infiltration and plant uptake. Removal efficiency of nitrogen is 72 %.

KEYWORDS

Cadmium removal; constructed wetland; domestic wastewater; eutrophication; nutrient removal; wastewater treatment.

INTRODUCTION

Constructed wetlands have been mainly designed to treat municipal and industrial wastewaters. They have common technologies in Europe, America and Australia where they are used widely for the tertiary treatment of domestic wastewater. In Europe they are particularly useful for polishing treated effluents to meet the stricter water quality discharge standards to the natural environment. They are also used increasingly for the treatment of low contamination wastewater. In the Netherlands, for example they are favored for the treatment of storm drainage run-off waters from highways and new housing estates. Industries chose them to clean up treated effluents to meet strict water quality regulations. The spread of the use of constructed wetlands to developing countries has been depressingly slow. Constructed wetland for domestic, non-toxic wastewater treatment in developping countries is a very powerful tool to raise the quality of life of local communities. The beauty of the system is the low technology, low capital costs and low maintenance required (Denny, 1997). Because of being constructed naturally and having no artificial process, eventually there remains no chemical and sludge. In addition wetland treatment is usually land intensive, a price paid for low input of non-renewable energy. They can be used also where eutrophication of fresh waters is a problem.

Thus around Lake Victoria (Which has an eutrophication problem, Balirwa, 1995) constructed wetlands as a polishing system for standard sewage treatment plant would be particularly helpful.

In this research Scirpus. spp plant (Figure 1) an integrated system of subsurface flow and surface flow is used and the feeding tank works as a presettling tank; the %30 of organics have been treated before discharging to the system. The parameters of, COD, BOD, TP, TKN, NO_3, Cd, SS, pH and temperatures were monitored. Because of working in semi-closed conditions real efficiency would be higher than observed. In condition of working with high loading rate or high water depth, in the sediment there becomes fast O_2 consumption and the treatment efficiency of BOD, N, P decreases.

Plant harvesting is needed for controlling the growth of algae and other components. If the wetland is not harvested the vast majority of the nutrients that have been incorporate in to the plant tissue will be returned to the water by decomposition processes.

Fig. 1 The appearance of constructed wetland system

Eutrophication is a particular threat in which species associations are changed, habitats lost and ecosystems destroyed. Therefore the appropriate controlling methods should be utilized. Most effective controlling methods of eutrophication in natural lakes or reservoirs is to obviate inlet of land-sourced nutrients, especially phosphorus. There occurs three methods for this problem:

1. Reducing phosphorus directly at source.
2. Treatment of stream inlet water.
3. To prevent wastewater-inlet to water source by collecting wastewater with sewage system and changing the direction of flow.

Getting high nutrient removal efficiency that could be interpreted, constructed wetlands are an appropriate clean-up method for eutrophication.

MATERIALS AND METHODS

Sampling and Analytical Procedure

The laboratory-scale experiment was performed within the campus laboratory of the Istanbul Technical University. Wastewater, primarily of domestic origin was pretreated in septic tank and then fed continuously two serials connected wetland cells. Inlets and outlets were sampled 3 days a week from June 96 to October

96. During this time, nutrients, organics and cadmium removal efficiencies were monitored. The parameters considered, were: Total phosphorus (TP), nitrate (NO_3^-), total kjeldahl nitrogen (TKN), chemical oxygen demand (COD). BOD, suspended solids (SS), pH and temperatures were rarely monitored according to the other parameters. All these parameters were determined as described in Standard Methods (American Public Health Association 1985.)

System Design

The system consisted of two identical serial cells.(Figure 2) The feeding tank has worked as a presettling tank. The wastewater has been given to the system with a flow of 5 l/d. At subsurface flow wetland generally the hydraulic loading rate(HLR) is 2-20 cm/d. In this system the HLR is 2.95 cm/d and the hydraulic retention time is 13 hours.

0.352 m length × 0.24 m width × 0.27 m depth; two serials connected cells were used . For obtaining hydraulic flow between the cells, the second was placed 11 cm lower than the first cell. The bed is backfilled with gravel sand mixture and planted with Bulrush (Scirpus. spp) . The characteristics of materials used in the system are shown in Table 1

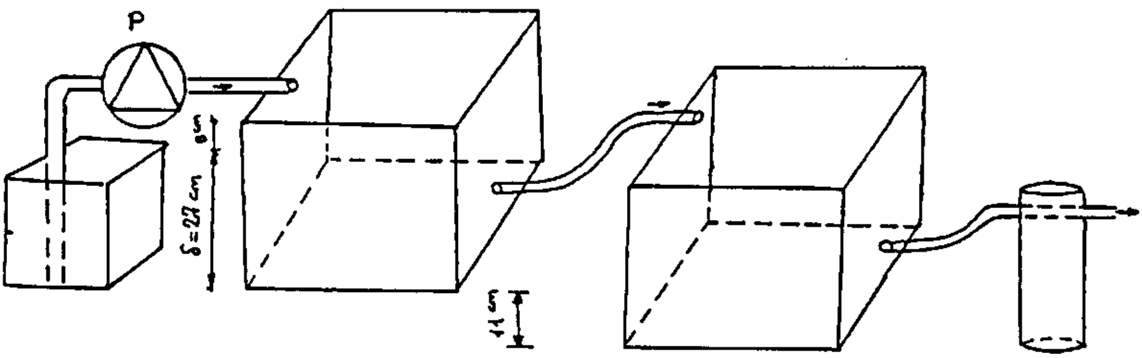


Fig. 2. The section of constructed wetland system

Table 1. The characteristics of materials used in the constructed wetlands

Material	The diameter of particle, mm	%
Sand	0.25	6-13
	0.5	80
	1.00	11
Gravel	8-19	-

Results and Discussion:

The wastewaters treated in the present study had characteristics of municipal wastewater . The influent wastewater was at constant levels during the experiments. The results presented here demonstrate the significantly reduce nutrients, organics and cadmium (Table 2), (Figure 3 and 4). According to the experiments it has been achieved 72% of TKN, 80 % TP, 75 % organics, 95% Cd removal efficiency.

Table 2 Influent-effluent concentrations and removal efficiency in the system

Parameters	Period	Average influent concentration mg/l	Average effluent concentration mg/l	Range of influent concentration mg/l	Range of effluent concentration mg/l	Average efficiency %
COD	June	271	87	-	33-76	68
	July	271.2	59	116-418	36-79	78
	August	207	72	80-500	53-97	65
	September	280	50.3	189-500	22-100	82
TKN	July	51.8	12	48-70	5-17	76
	August	47	9	44-52	4-14	80
	September	56	14.5	40-72	12-19	72
	October	59	24	52-767	22-25	58
TP	July	5.69	2.525	4.3-10	1-3.6	51
	August	6.86	2.154	4.5-9	1-3.5	69
	September	9.43	3.23	7-12	1.7-4.7	66
Cd	August	0.422	0.025	0.2-0.91	0.01-0.0425	94
	September	0.608	0.0278	0.4-0.895	0.025-0.03	95

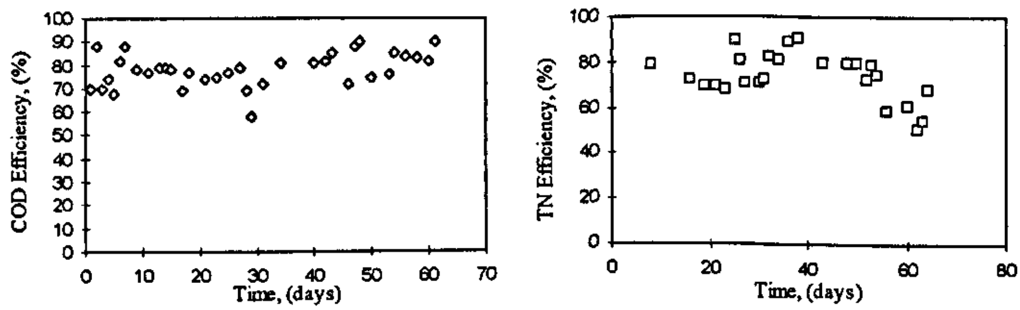


Fig.3. COD and TN Efficiency

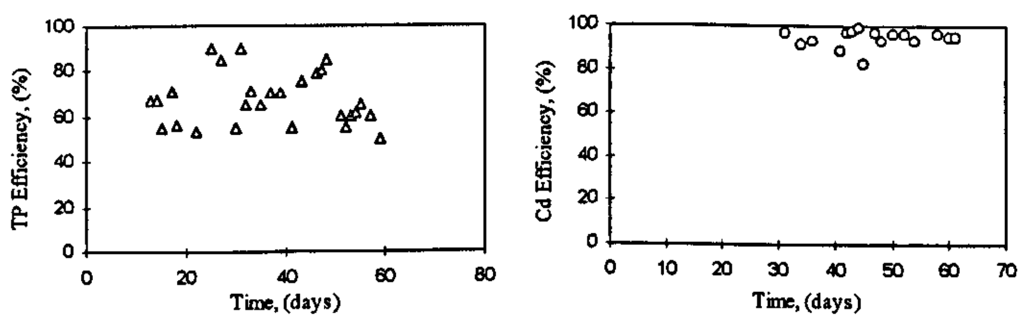


Fig.4. TP and Cd Efficiency

Nitrogen compounds removal in wetland is governed mainly by microbial nitrification and denitrification , whilst other mechanisms such as plant uptake , assimilation and ammonia volatilization are generally of less importance in the nitrification process ammonia is oxidized mainly to nitrate. The nitrate is subsequently reduced to nitrogen gas in the denitrification process , where biomass or other organic residues presented in the wetland are used as the carbon and electron source (Reed and Brown, 1995). In the influent total nitrogen was approximately equal to TKN, in the effluent was equal to TKN + NO_3-N.

The removal of TP realizes by these mechanisms : - Sedimentation and burial via plant uptake.- Adsorption on to the substratum. - Precipitation and complexation reactions and biological incorporation into biological film.(Richardson, 1985) These biotic and abiotic processes regulate phosphorus removal by wetlands.

Organic matter may be removed by (1) settling of particulate matter. (2) Breakdowns of soluble organics during microbial respiration(aerobic and anaerobic catabolic activity of heterotrophic microorganisms) Litter and sediment decomposition produce soluble carbon compounds which are background organics in the system. Consequently for obtaining high removal efficiency HLR should have high values (Tanner et al, 1995)

The aim is to assess the removal of Cd within a lab-scale wetland and the role of the substrate, the root zone and subsurface tissues of the macrophytes in metal uptake. The results are lower than the maximum Cd concentrations. The removal efficiency of Cd may decrease if the system becomes saturated with heavy metals (Mungur et al, 1997).

Site characteristics that must be considered in wetland system design include topography, soil characteristics, existing land use, flood hazard and cilimate : -Level to slightly sloping, uniform topography is preferred for wetlands systems. -Sites with slowly-permeable surface soils or subsurface layers are most desirable for wetlands systems. -In general, wetlands sites should be located outside of flood plains.

Vegetation places an integral role in wetland treatment by transferring oxygen through their roots and rhizome systems to the bottom of treatment basins and providing a medium beneath the water surface for the attachment of microorganisms that perform most of the biological treatment.

The objectives of most wetlands pollution control projects are : (1) Water quality and enhancement. (2) Water storage and flood attenuation? for storm water project. Secondary benefits (Photosynthetic production; secondary production of fauna; food chain and habitat diversity; export to adjacent ecosystems; aesthetic, recreational, commercial and educational human uses.)

CONCLUSIONS

The following conclusions may be drawn in relation to the biological activities in wetlands systems: In wetland treatment system; the rooted, hydrophilic plant species and shallow, saturated soil should be used. Constructed wetlands systems can be used at small residential and touristic areas and the place where the lands are cheap and large. They are alternative systems for increasing the quality of water taken after secondary treatment. According to the data set by European Community; the results achieved by constructed wetlands have great efficiency on removal of most pollutant parameters. In the future, wastewater treatment efficiency variability is thought to be observed by setting laguna system fore and back of the wetland. Besides it is planned to set the system with a combination of floating and rooted plants.

ACKNOWLEDGEMENT

The authors are pleased to acknowledge the support of Scientific and Research Council in Turkey and Assoc. Prof. Lütfi Akça and Dr. Hürrem Bayhan for their valuable advises.

REFERENCES

Denny, P. (1997). Implementation of constructed wetlands in Developing Countries. *Wat. Sci. Tech.*, Vol. 35, No 5, pp 35-43

Kadlec, R. H. and Knight, R. L. (1996). *Treatment Wetlands*. CRC Press.

Tanner, C. C., Clayton, J. S. and Updell, M. P. (1995). Effect of loading rate and planting on treatment of daily form wastewaters in constructed wetlands. Removal of oxygen demand, suspended solids and fecal coliforms. *Water Res.*, 291(1), 17-26

Mungur, A. S., Shutes, R.B.E., Reuitt, D. M. And House, M. A. (1957). An assessment of metal removal by a laboratory scale wetland. *Wat. Sci. Tech.*, Vol. 35, No 5, pp. 125-133

Reed, S. C. And Brown, D. (1995). Subsurface flow wetlands a performance evaluation water. *Env. Res.*, 67(2), 244-248

Richordson, C. J., (1985). Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science*, 228, 1424-1427

Tchobanoglous, G. and Burton, F. L. (1991). *Metcalf & Eddy Wastewater Engineering, Treatment-Disposal-Reuse*, pp. 992 - 1002

MANAGEMENT AND POLICIES

Extended summary

Which policies can stop large scale eutrophication?

by

Curt Forsberg, Institute of Limnology, Uppsala university, Uppsala, Sweden

Large scale eutrophication of fresh and marine waters is the result of increasing discharges of phosphorus (P) and nitrogen (N) from modern society. The major causes for this increase is the growing world population, from 3 to 6 billion people since 1960, and changed consumption pattern of P and N in the industrialised parts of the world. The doubling of the world population has been facilitated by advanced technology for improving land productivity, including several-fold growth in use of fertilizers, the basis for the increase of the world food output. Increased application of fertilizers (incl. manure), increased irrigation, and a growing intensive animal production have given rise to substantial nutrient losses to the waters, so-called diffuse or non-point source pollution.
From agricultural production there has been an increasing export of crops (incl. nutrients) to urban areas. These areas have expanded and are expected to continue to grow. More and more nutrients have been channelled through the food chain into the sewage systems, and have been discharged through pipes, so-called point sources.
As science has identified P and N as the key elements in man-made eutrophication of fresh and marine waters, present policy interventions are focused on the reduction of the loading of these elements.

In lake eutrophication the interest has primarily been focused on P removal from municipal sewage and by substituting phosphate in laundry detergents with other non-fertilizing compounds. Different policies have been used; from P removal with best available technique applied uniformly across a whole country, e.g. in Sweden, to defined effluent standards introduced selectively in certain catchments with evident eutrophication problems. Advanced waste water treatment for P removal has improved many lakes, at the same time as huge amounts of P enriched sludge has been distributed in overdose in agriculture or stored in landfills. The size and rate of future leakage of P from these P accumulating systems are unknown.

The increasing eutrophication of marine waters has resulted in a focused interest to reduce N from agriculture, and recently from municipal sewage. In the EC commission there are concerns about the negative impact on the waters of the present animal husbandry practicies and the intensive use of fertilizers (the common agriculture policy, CAP).

Policy interventions discussed and applied are:
- legislation to control application of animal manure as in Denmark, France, Germany, the Netherlands, and Belgium.
- economical, including statutory government interventions such as bans and regulations, and market and price oriented instruments, e.g., taxes and quotas. Governmental declarations (in Sweden and Denmark) to reduce the discharge of N has not been successful so far.

There are relationships between the size of the world population and the consumption of raw phosphate, and between the population density in the watershed and export of phosphate and nitrate in the world's major rivers. In order to control eutrophication as well as other large scale environmental problems, there is a strong need of policies for birth control. But, as is well known, a number of socio-economical and religious barriers make birth control very difficult to perform. This means that the driving force for eutrophication, the food production for the human being and the sewage and sludge after consumption will increase during the coming decades, as the world population continues to grow.
There is also a strong need of policies which can improve recycling of nutrients between urban and rural areas. Planning policies must include a higher degree of integration between countryside and urban settlements. However, the prognosis shows a further increase in urbanization, from about 40% in 1990 to 60% in 2025, which counteracts effective recycling. The growing world population in combination with further increased urbanization counteracts the possibility to decrease eutrophication of our waters. In addition, losses from already existing storages of nutrients in nutrient saturated soils, in sediments, in sludge, ashes and organic household wastes in landfills can be expected to operate in the same direction. Climate changes are also expected to increase eutrophication in some areas.

While awaiting methods to minimise the losses of nutrients in agriculture and effective systems for nutrient recycling, strong policies must be developed which can help the society not to waste N and P. All laundry detergents should be phosphate free, all phosphate additives in foods and beverages should be eliminated, and the over-consumption of N by eating to much animal protein should be stopped. By decreasing the intake of P (phosphate additives) and N (animal protein) one can expect a reduction of the nutrient loading on the waters with thousands of tons.

Which policies can then stop large scale eutrophication? This question is at the moment impossible to answer. Policy making and implementation must result in minimal losses of N and P from agriculture, and maximal recycling of these nutrients between urban and rural areas. There is a number of barriers to overcome, but also a lot of space for innovations.

COMPREHENSIVE APPROACHES TO WATER MANAGEMENT

Peter T.J.C. van Rooy*, Joost de Jong**, Eric Jagtman***,
S. Harry Hosper ****, Paul C.M. Boers****

* DHV Water BV, P.O. Box 484, 3800 AL Amersfoort, The Netherlands.
** Ministry of Transport, Public Works and Water Management and Technical University Delft, P.O. Box 5048, 2600 GA Delft, The Netherlands.
*** National Institute for Coastal and Marine Management, P.O. Box 20907, 2500 EX The Hague, The Netherlands.
**** Institute for Inland Water Management and Waste Water Treatment, P.O. Box 17, 8200 AA Lelystad, The Netherlands.

ABSTRACT

For us to have sufficient amounts of good quality fresh water available in the 21st century, it is necessary that we find a different way of dealing with water. Awareness of this fact represents a first step in the right direction. Interactive planning can substantially contribute to this awareness and therewith to the solution of both physical as well as cultural bottlenecks. Interactive plan processes are complex. The challenge lies in being actively involved in this complexity. The IPEA methodology has been developed to support adequate completion of interactive plan processes. This methodology has already been applied in a number of plan processes, 25 of which were recently evaluated. From this evaluation it appears that applying IPEA helps to increase the clarity and the suitability of both processes and plans. The methodology also makes a positive contribution to levels of communication and interaction between the people and organisations involved in a plan process. Furthermore, IPEA leads to an increased acceptance of the contents of plans.

KEYWORDS

Comprehensive water management; sustainable development; interactive planning; participatory decision making; IPEA.

INTRODUCTION

For a number of years now the already existing but ever-more threatening shortage of quality fresh water has been an issue of global importance. A World Bank prognosis suggests that more than half the worlds' population will encounter a shortage of good quality water in the next century (Saeijs, 1995). This is a direct consequence of the way we currently deal with water. In Tunisia, for example, groundwater is being extracted ten times faster than it can be replaced with rain water. Another example is the use of groundwater in and around Beijing: between 1950 and 1997 the groundwater level has dropped by as much as 45 metres (Terpstra, 1997), a drop of one metre per year! The seriousness of these developments explains the substantial annual rise in publications regarding this subject. Similarly, many people explicitly pointed out the necessity for a change in the way we deal with water during the 'Living with Water' (1994) and 'Future Water Quality Management in Europe' (1996) congresses (De Jong et. al., 1995; Jagtman, 1997).

According to many experts, dealing differently with water is necessary in order to tackle physical as well as cultural bottlenecks. Examples of physical bottlenecks are eutrophication, pollution and desiccation. Examples of cultural bottlenecks are the excessive use of resources and inadequate co-operation both between water managers among themselves and between water managers and other organisations involved in water management. The attention given to cultural bottlenecks has risen enormously, especially in recent years. On a world-wide scale we seem to have reached the stage where we increasingly connect our own role and attitude to the quality of our environment.

An historical perspective is essential for any comprehensive understanding of why a different approach to dealing with water is so necessary. This article gives a sketch of developments between prehistory and the year 2025. In addition, it indicates just what this new approach consists of: a comprehensive approach towards water management projected onto the beginning of the next century. To reach such a comprehensive approach requires targeted planning towards this goal. The emphasis will have to shift from planning between four walls to an open, process-like planning. It no longer involves just fleshing out the setting of tasks of one organisation directed at one management area. Instead it requires interactive planning and decision making directed at a water system or catchment area which has to be mutually managed (Van Rooy, 1996).

WATER MANAGEMENT UNTIL THE PRESENT TIME, WITH EMPHASIS ON THE NETHERLANDS

One thing about water is certain - it has been around on our planet for billions of years. For virtually all of this time water systems existed without the involvement of a single human being. The existence of the current human being (*Homo Sapiens*) can be dated to around 40,000 years ago. Until approximately 3,000 years ago the use of water by these people had almost no consequences for water systems.
Population density was low. Man lived in a certain harmony with his environment and any pollution was dealt with by the self-cleansing ability of water. A natural equilibrium existed which had no need for water management. But around 3,000 years ago this situation began to change in Europe (especially in southern areas). A large increase in and greater concentration of the population meant the user-capacity of the physical surroundings was exceeded and the necessity to deploy techniques arose. Thus came the first provisions for drinking water and sewerage as well as hydrological interventions aimed at, for example, agriculture. This stage amounted to a basic form of water management, one which lasted in Europe until about 200 years ago or, in other words, five or six generations back. From that time onwards the population increased rapidly. The number of inhabitants in The Netherlands, for example, has risen from five million in 1900 to the present total of 16 million. This led to an exponential increase in the use of water and space and the large scale deployment of techniques to control water systems. Furthermore, many organisations dealt with water from specific desires and points of view. Unsurprisingly, therefore, interventions in water systems by different organisations were frequently unconnected and often even contradictory. This sectoral approach to water systems led to problems such as eutrophication and pollution - bottlenecks which were also tackled in a sectoral way. As far as eutrophication is concerned, for example, this led to international agreements about banning phosphates from washing powder and the extensive purification of waste water (Hosper, 1997). Despite such interventions (which are very important for water) the bio-diversity of water systems continued to decrease and problems with water quantity also began to be revealed: desiccation, even desertification, while elsewhere floods occur.

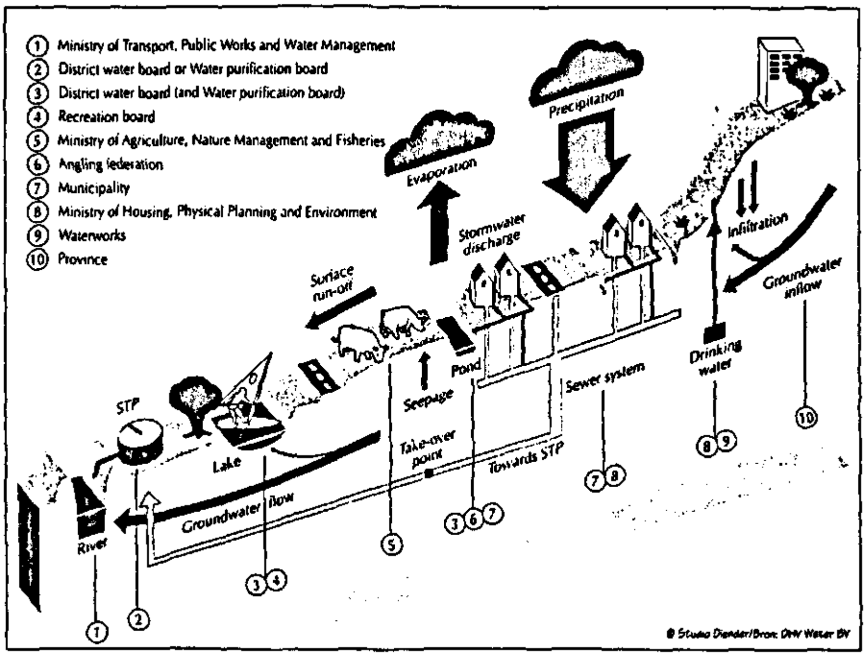
Around ten years ago it became clear that continuing a sectoral approach to water systems would have catastrophic consequences within a few decades. The course had to be changed. Instead of a sectoral approach, water systems had to be approached integrally. From this approach it became apparent that water systems consist of water, sediments and banks and that a connection exists between ground and surface water and water quantity and quality. In regard to the use of water the necessity became apparent of selecting certain areas for certain types of use. Furthermore, it underlined the fact that water is a part of our space and, therefore, that policies for water are also concerned with policies in the field of physical planning and environment, and vice versa.

The body of thoughts behind this integrated approach towards water systems was adopted globally in just a few years.

Despite this swift adoption, however, the implementation of integrated water management has lagged behind the theory. What has been brought into practice is in most cases connected to attempts to recover affected ecological functions. Examples include the removal or treatment of polluted soil sediments, the construction of environmentally-friendly banks, and the replenishing of fish stocks (Boers, 1991; Hosper, 1997). In The Netherlands these reparative measures have led to a reversal of trends in the development of the bio-diversity. Instead of a further decrease in the number of different kinds of plants and animals in and around the rivers, there has now been a slight increase. Quite apart from these measures, however, new problems are already on the horizon. After pollution, eutrophication, desiccation, flooding, etc. comes issues such as the emission of materials which influence the hormonal balance. There are clear indications that such materials, which are absorbed through (drinking) water, can cause endocrinal disturbances in people and animals (Leonards et. al., 1996).

Research into the reasons for the failure of the (in broad terms) integrated approaches to water systems points to the existence of many bottlenecks: both physical as well as cultural. Physical bottlenecks are concrete and tangible. As previously indicated, they can revolve around eutrophication and desiccation but may also involve the traditional water-managerial organisation of urban areas and the sectorally-targeted water management of rural areas. Cultural bottlenecks are, broadly speaking, connected to the morality of how we deal with water and other resources, with the institutional division of tasks and responsibilities regarding water management (see figure 1) and with the communication between the organisations and people involved, directly or indirectly, in water management (Van Rooy, 1995). There is a clear relationship between physical and cultural bottlenecks. Put in black and white terms, current physical bottlenecks are a consequence of past cultural bottlenecks. Thus current cultural bottlenecks will lead to future physical bottlenecks.

Figure 1 A cross-section of a fictitious area with an indication of the organisations involved in water policy and/or management in The Netherlands.



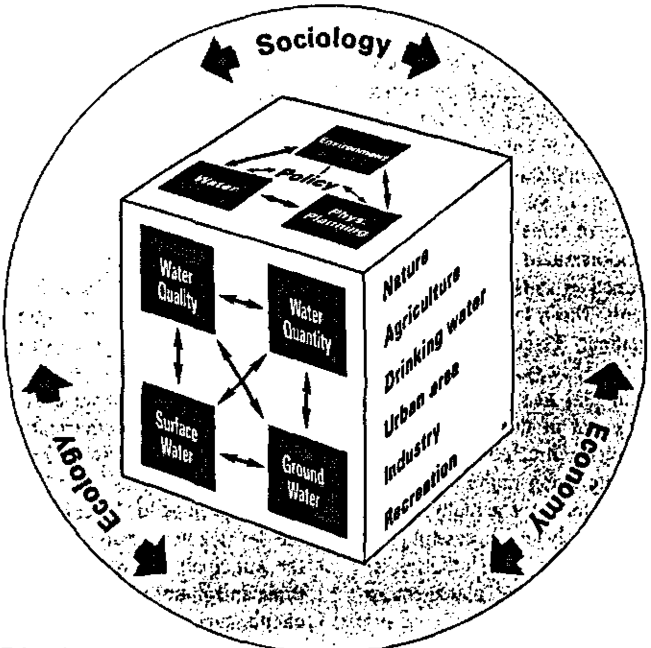
Solving bottlenecks requires, at the very least, an awareness of the way we deal with water. Implementing tangible demonstrative projects could substantially contribute to this by making civilians aware of the difference between traditional and progressive ways of dealing with water. Several projects in new residential areas in The Netherlands have in the meantime shown the potential of almost closing nutrient cycles in urban water systems (De Jong, 1996). Helped by the enthusiasm shown by civilians, these projects are being replicated in newly-built residential and existing urban areas. Awareness of the relationship between the way in which we deal with water and the available quantity of good quality water will also be necessary in order to explain the need for large investments in the short term. A recent study has shown that any real step forwards in water management in The Netherlands between now and 2015 requires an investment of more than 34 billion guilders (Ministry of Transport, Public Works and Water Management, 1996).

Any failure to provide sufficient investment now will see even larger investments having to be made in the future. A solid grounding of water management within society also means that the many organisations involved in water management are forced to give up their protected position, no longer forming together a separate sector within society but interacting with other social fields of influence. Thus seen, the integrated water management stage can be characterised as a phase of awareness; a phase which does not simply work a little less sectorally but involves a genuine change of course. Our mutual future requires a comprehensive approach towards water management (Van Rooy & De Jong, 1995).

TOWARDS COMPREHENSIVE APPROACHES

In 1987 the World Commission on Environment and Development depicted what can be considered as a comprehensive approach (Brundtland, 1987). As far as water management is concerned it means becoming fully involved with three decisive social fields of influence - ecology, economy and sociology, each of which is closely related to the others (see figure 2). Ecology represents all inorganic sources, organic sources and living organisms, with their mutual relationships. Economy stands for the way that human potential is used and the methods employed to utilise inorganic sources, organic sources, flora and fauna. Sociology means the way in which people interact with each other in relation to the consequences this has for ecology and economy. Water management increasingly forms an essential link between these three fields. Awareness of this means that water management can no longer be a case solely for directly-responsible organisations but for society as a whole.

Figure 2 Comprehensive water management: integrated water management (link between water systems, water uses and policy fields) set within an interactive framework of ecology, economy and sociology.



The following outlines what a comprehensive approach to water management entails from the perspective of the three fields of influence.

Ecology

- Although water and land are recognisable entities within the physical space, they are closely linked. Water, for example, can be considered to be the designer of land and land can be considered to be the host for groundwater. Given this connection, good water management requires a total approach towards the physical space. In other words, catchment areas need to be taken as a starting point if water is to be approached as a system. Only then can connections be made between the use of land and the functioning of water systems. The importance of this can be illustrated by looking at the development of the eutrophication of Dutch waters. As a consequence of the "Rhine and North Sea Action Programme" the phosphorus concentration in both the Rhine as well as the North Sea decreased by more than half. This also applies to the large fresh water systems that form part of the Rhine basin (De Vries et. al., 1997). For a lot of smaller fresh waters, however, which are far more under the influence of diffuse sources, excessive algae growth is still relatively normal. The only chance for these waters is a combined approach towards water and land.
- A comprehensive approach to the physical environment also has consequences for (applied) scientific research into the functioning of water systems. As far as eutrophication is concerned, for example, it is no longer so much an issue of the relationship between concentrations of one nutrient and the density of one kind of algae. Research will have to focus much more on the effects of changed concentrations of nutrients on the functioning of relationships between abiotic and biotic elements of ecosystems.
- A comprehensive approach does not just involve space but also time. After all, everything occurring now has its origin in the past and is making its mark on the future. By creating temporal relationships it is possible to anticipate desired or undesired developments. Thus it is useful for water systems research in The Netherlands to take into account rapid urbanisation and a rise in sea level.

Economy

- The number of forms of use has increased over the years. Whereas in the distant past only drinking water and water for agriculture were needed, today it is in demand as process and cooling water for industry, for shipping purposes, in the many forms of recreation, and in countless other ways. Good quality water has become of increasing economic importance.

The following outlines what a comprehensive approach to water management entails from the perspective of the three fields of influence.

Ecology

- Although water and land are recognisable entities within the physical space, they are closely linked. Water, for example, can be considered to be the designer of land and land can be considered to be the host for groundwater. Given this connection, good water management requires a total approach towards the physical space. In other words, catchment areas need to be taken as a starting point if water is to be approached as a system. Only then can connections be made between the use of land and the functioning of water systems. The importance of this can be illustrated by looking at the development of the eutrophication of Dutch waters. As a consequence of the "Rhine and North Sea Action Programme" the phosphorus concentration in both the Rhine as well as the North Sea decreased by more than half. This also applies to the large fresh water systems that form part of the Rhine basin (De Vries et. al., 1997). For a lot of smaller fresh waters, however, which are far more under the influence of diffuse sources, excessive algae growth is still relatively normal. The only chance for these waters is a combined approach towards water and land.
- A comprehensive approach to the physical environment also has consequences for (applied) scientific research into the functioning of water systems. As far as eutrophication is concerned, for example, it is no longer so much an issue of the relationship between concentrations of one nutrient and the density of one kind of algae. Research will have to focus much more on the effects of changed concentrations of nutrients on the functioning of relationships between abiotic and biotic elements of ecosystems.
- A comprehensive approach does not just involve space but also time. After all, everything occurring now has its origin in the past and is making its mark on the future. By creating temporal relationships it is possible to anticipate desired or undesired developments. Thus it is useful for water systems research in The Netherlands to take into account rapid urbanisation and a rise in sea level.

Economy

- The number of forms of use has increased over the years. Whereas in the distant past only drinking water and water for agriculture were needed, today it is in demand as process and cooling water for industry, for shipping purposes, in the many forms of recreation, and in countless other ways. Good quality water has become of increasing economic importance.

For example, the growth of algae in the Mediterranean - whether poisonous or not - leads annually to a serious loss of income for the tourist trade. Preventing algae from growing in this case has an economic value and can be expressed in financial terms. With that, a more integrated cost-benefit analysis of economic activities can be stimulated. This can place economic activities in a different light (Hueting, 1992). Import of feed for intensive cattle breeding, for example, will seem a much less suitable option based on an integrated cost-benefit analysis than on a sectoral one. After use, this feed plays an important part in the increase of phosphorous and nitrogen concentrations in water systems. It is obvious that a comprehensive approach to water systems benefits from integrated lost-benefit analysis.

- The growing complexity of dealing with water has also led to an increase in the variety of possible measures. In addition to technical interventions, activities that influence the behaviour of civilians have also come into the picture. Facing this rapid rise in possibilities lies a budget that, in general, is growing at a much slower pace. This calls for the setting of priorities - the social importance of setting the right priorities justifies a well-considered and perceptive process of prioritisation. At least one criterion is necessary to achieve this, the closest to hand being the cost-effectiveness of the different possible measures. But in this world of ours wishes play equally an important role as facts. For this reason it is preferable that a second criterion, the level of support for the implementation of measures, is involved in the prioritising. In this way a traceable and accepted choice of measures can be made, the implementation of which will in return be given the highest priority.
- In connection to the previous point, legally prescribing generic norms is not desirable in all cases. It could lead to a disproportionately expensive intervention in order, for example, to remove the last bit of phosphorous from effluent. Similar means may well be better utilised for other measures which have far more effect on improving the functioning of water systems. No longer prescribing generic standards, however, does not mean that standards related to a basic level or maximum permissible risks can be abandoned. It certainly does mean that extra efforts are linked to specific regional circumstances in both an ecological and economic respect as well as sociologically.

Sociology

- The world's population is growing faster than ever before: 1950's 2.5 billion figure will have risen to 6.3 billion by the year 2000. All these people require space, yet the available habitable earth surface remains virtually the same.

Presuming the current percentage growth of the world's population, by the year 2550 there will be only one square metre of habitable land per person as opposed to today's two hectares (Teunissen, 1989). And this calculation takes no account of the fact that people also want more individual space.
No further demonstrations are required of the fact that pressure on space is already great and will become even larger. What applies to space, goes for water too. Considering the overall importance of having enough good quality water available, water management has become an issue for society as a whole and can no longer remain the domain of organisations that are mainly concentrated on what the possibilities are in a technical respect.

- As the number of people has continued to increase, so has the emancipation of civilians, especially in Western countries. This has resulted in civilians wanting to become involved in those planning and decision making processes which are important to them. This need for participation means it is no longer sufficient to draw up a plan behind closed doors and then present it to society as an accomplished fact. Civilians are demanding insights into the steps that were completed during a plan process on the one hand and the chance to actively contribute to it on the other. This in turn means that the traditional reactive planning is making way for interactive planning.
- The close relationship between land and water, the crucial importance of good water management and the increasing pressure on space and water all have influence on the scope of planning. Whereas in the recent past it was still sufficient to have a plan process based on the setting of tasks of one organisation and directed at one management area, now we will increasingly see spatially integrated plan processes at the level of, for example, a catchment area or parts thereof. Judicial-managerial borders will preferably hardly ever play a role in this: after all, using the most suitable available means is in our mutual future interest.

INTERACTIVE PLANNING

As previously indicated, developments in water management mean a new approach towards planning is required. Thus the quality of the plan process is equally as important as the plan resulting from that process, there will have to be more room for participation by interest groups and civilians, and water will have to be viewed more as part of the physical space. This means that the complexity of planning radically increases in comparison with the more sectoral and closed planning: the challenge lies in suitably handling this increased complexity. More insight into planning is necessary to achieve this. Within planning three elements can be distinguished: contents, process and structure (Van Rooy, 1996).

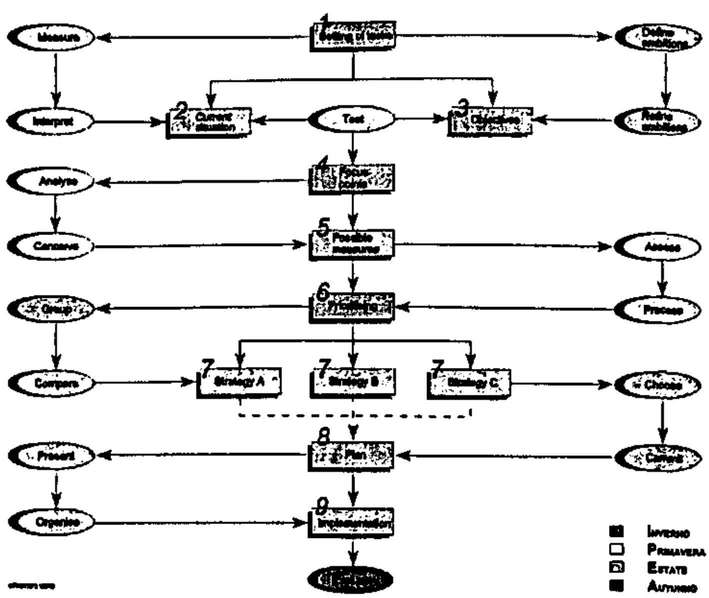
The contents are, in the context of this article, related to one or more water systems. The process is on the one hand dependent on the points of departure and the set preconditions and on the other on the players in the field or, in other words, those people involved in the process. The latter in particular make each plan process by definition unique. The structure can be looked upon as the language which is spoken and the grammar which is employed. In contrast with the contents and the process, the structure is not necessarily unique for each plan process. On the contrary, the structures of plan processes appear, in general, to have more similarities than differences. The differences manifest themselves on the one hand in the use of separate definitions for the same things or in the use of the same definitions for different contents and on the other hand in the sequel of steps that are to be followed. These differences are not essential for the uniqueness of plan processes but do, however, create an obstacle when fine tuning during plan processes or completing plan processes in an interactive way. Agreements about the meaning of definitions and a clear structure could remove these drawbacks and could also serve as a footing during planning. In short: a clear structure acts as a framework for the unique contents and the unique process. Communication between people and organisations can be improved and all attention can be focused on the contents and the process.

Recent decades have seen various system-analytical studies implemented into realising a logical and workable structure for plan processes. These include, for example, the Policy Analysis of Water Management in The Netherlands (PAWN) and studies by Walker into steps in the policy analysis process (Pulles, 1985; Walker, 1988). These and similar studies have approached plan processes mainly in a technocratic way and have resulted in structures which are related to traditional plan processes. With these structures as a starting point, a structure for interactive plan processes has been developed. The structure consists of nine steps and 16 related activities (see figure 3). The steps are on the one hand based on the results of previous activities and on the other serve as the basis for future activities. Fleshing out the activities involves one or more groups of people, such as politicians, policy makers, managers, interest groups and civilians. Tools have been developed with an eye to adequately completing the structure: three methodologies and one approach. In conjunction with the cyclical character of plan processes these have been named after the four seasons. The free association between seasons and the works of the musician Vivaldi were responsible for these seasons being specified in Italian: INVERNO (winter), PRIMAVERA (spring), ESTATE (summer) and AUTUNNO (autumn). Applying INVERNO results in an overview of focus points on which a future plan period should be directed (Van Rooy et. al., 1996). Applying PRIMAVERA leads to prioritising all possible measures based upon assessment of the cost-effectiveness and the support (Van Rooy et. al., 1997a).

Applying ESTATE results in the choice of a strategy, an intrinsic completion of the previous plan process and serves as the basis for implementation of the plan (Van Rooy et. al., 1997b). The AUTUNNO approach is related to the evaluation of the plan process.

Although it is possible to apply these methodologies separately, an integrated application seems more obvious. The three methodologies and the approach are collectively known as the IPEA methodology, the acronym for the four seasons.

Figure 3 A schematic depiction of the IPEA methodology as a whole of the steps, activities and mutual relations within a plan process. The steps indicated in the squares correspond to a widely applicable structure for planning. The activities, indicated in the ovals, correspond with the fleshing out of the steps. Because of iterations within a plan process many other connections are feasible, but for reasons of clarity these have not been included in the figure.



IPEA has been (completely or partially) applied in a number of plan processes, including that aimed at the restoration plan for the River Vecht and the strategic policy document on the reconstruction of water and sanitation in Bosnia Herzegovina. Some 25 plan processes have recently been evaluated (Van Rooy et. al., 1997c) with the main focus being on clarity, suitability, communication, interaction and acceptance.
As far as clarity is concerned, the value of IPEA seems to lie in the requirement to explicitly complete each step and activity. This has led to greater insight into the connections between the different steps in virtually all of the plan process evaluated. Surveys seem to indicate that IPEA's contribution to increasing the suitability of the evaluated plan processes has, on balance, been a mildly positive one. This careful qualification is probably related to a number of factors: that only a part of IPEA has been applied in almost all plan processes, that the decision to do so was only taken at a later stage, that concept versions were used out of sheer necessity and that there is virtually no question of implementing the formulated plans. As far as communication is concerned, in general IPEA appears to contribute in a positive way to the exchange of information, knowledge and points of view. In terms of interaction, applying IPEA appears to positively contribute to mutually influencing those people involved in plan processes. As far as acceptance of plans is concerned, IPEA appears to favourably contribute to plan realisation; a plan which is seen by the people involved as resulting from mutual labour and, as a consequence, particularly feasible.

In addition to these results the evaluation showed that it is not IPEA itself but the way that it is put into practice that is decisive for the way in which water systems are being approached. In other words, the mere fact of applying IPEA does not lead to the transformation of a traditional approach into a progressive approach towards water systems.

IN CONCLUSION

- World-wide eutrophication is only one of the physical bottlenecks in water management. Other physical bottlenecks are pollution with xeno-biotic materials, contamination from sediment and solid materials, desiccation, desertification, flooding, etc. The way to tackle these bottlenecks is to take a different approach to water. To achieve this demands a genuinely radical change of course.
- In addition to the existence of physical bottlenecks, the fact that cultural bottlenecks also exist is being increasingly recognised. Solving these requires among other things giving water systems a much higher priority in our thoughts and actions.

In other words, approaching water not only in terms of usage but also as a source, one that has its own patterns.

- To deal in a substantially different way with water requires an increasing socialisation of water management, whereby it starts to be seen as an essential link between three decisive social fields of influence - ecology, economy and sociology. It also requires a comprehensive approach in terms of space as well as time.
- It becomes more and more important for water management that the costs and benefits of interventions are considered in an integrated fashion, that priorities are set based upon cost-effectiveness and support and that realistic and area- specific norms are employed.
- For water management to be supported and well thought-through requires increasingly interactive planning whereby interest groups and civilians can participate during plan processes and whereby water is being considered a part of the physical space.
- The complexity of interactive planning on the one hand and the necessity for good communication on the other means agreements about the structure of plan processes are required.
- Building on existing system-analytical knowledge, a uniformly-applicable methodology for interactive plan processes has been developed. The methodology is called IPEA and creates a pattern of thought for completing an entire plan process. The pattern of thought will have to be fleshed out in more detail for each individual plan process. Although applying the methodology supports (complex) plan processes, it offers no guarantee for a progressive approach to water systems. That continues to depend on the actual users of the methodology.
- The evaluation of IPEA's applications indicates an increase in the clarity and traceability of plan processes, a rise in the suitability of plan processes, improved communication and interaction between people and organisations and a broader acceptance of the contents of plans.

REFERENCES

Boers, P.C.M. (1991). The release of dissolved phosphorus from lake sediments. *Thesis Wageningen Agricultural University.*

Brundtland, G.H. (1987). Our common future. *Oxford University Press.* Oxford, New York. ISBN 0-19-282080-X.

De Jong, J., Van Rooy, P.T.J.C. & Hosper, S.H. (1995). Living with water at the cross-roads of change. *Water Science and Technology 31 (8).*

De Jong, J. (1996). Waterbeheer in de komende eeuw. *Natuur & Techniek (65) 6* (in Dutch).
De Vries, I., Duin, R.N.M., Peeters, J.C.H., Los, F.J., Bokhorst, M. & Laane, R.W.P.M. (1997). Patterns and trends in nutrients and phytoplankton in Dutch coastal waters: comparison of time series analysis, ecological model simulation and mesocosm experiments. *RIKZ*, The Hague.
Hosper, S.H. (1997). Clearing Lakes. An ecosystem approach to the restoration and management of shallow lakes in the Netherlands. *Thesis Wageningen Agricultural University.*
Hueting, R. (1992). Economic evaluation of environmental impacts. Design of a cost benefit analysis of the Lower Guayas flood control project, Equador. *The Environmentalist 12 (2).*
Jagtman, E. (1997). EWPCA-NVA conference on future water quality management in Europe. *European Water Pollution Control 7 (1).*
Leonards, P.E.G., Vethaak, A.D. & De Voogt, P. (1996). Oestrogeen-actieve stoffen in het milieu. *Rijksinstituut voor Kust en Zee*, The Hague. ISBN 90-369-0415-3 (in Dutch).
Ministry of Transport, Public Works and Water Management (1996). Toekomst voor Water. Lelystad, The Hague. *RIZA Nota 96.058, RIKZ Nota 96.030.* ISBN 90-36-95-0341 (in Dutch).
Pulles, J.W. (1985). Beleidsanalyse voor de waterhuishouding in Nederland/ PAWN. *Rijkswaterstaat*, The Hague. ISBN 90-12-05184-3 (in Dutch).
Saeijs, H.L.F. (1995). Dorstige aarde. Inaugurele rede. *Erasmus Universiteit Rotterdam* (in Dutch).
Terpstra, G. (1997). Beheer van grondwater. *Proceeding within the framework of a conference on integrated water management at the University of Brabant* (in Dutch).
Teunissen, D. (1989). Aan alles komt een einde. Exaugurele rede. *Katholieke Universiteit Nijmegen* (in Dutch).
Van Rooy, P.T.J.C. & De Jong, J. (1995). Towards comprehensive water management in The Netherlands (1): developments. *European Water Pollution Control 5 (4).*
Van Rooy, P.T.J.C. (1995). Towards comprehensive water management in The Netherlands (2): bottlenecks. *European Water Pollution Control 5 (6).*
Van Rooy, P.T.J.C. (1996). Towards comprehensive water management in The Netherlands (3): planning. *European Water Pollution Control 6 (3).*
Van Rooy, P.T.J.C., Van Sluis, J.W., Tolkamp, H.H. & De Jong, J. (1996). Towards comprehensive water management in The Netherlands (4): INVERNO. *European Water Pollution Control 6 (6).*

Van Rooy, P.T.J.C., Van Sluis, J.W., Tolkamp, H.H. & De Jong, J. (1997a). Towards comprehensive water management in The Netherlands (5): PRIMAVERA. *European Water Pollution Control 7 (1).*

Van Rooy, P.T.J.C., Van Sluis, J.W., Tolkamp, H.H. & De Jong, J. (1997b). Towards comprehensive water management in The Netherlands (6): ESTATE. *European Water Pollution Control 7 (5).*

Van Rooy, P.T.J.C., Van Sluis, J.W., Tolkamp, H.H. & De Jong, J. (1997c). Towards comprehensive water management in The Netherlands (7): AUTUNNO. *European Water Pollution Control 7 (6).*

Walker, W.E. (1988). Generating and screening alternatives. *In: H.J. Miser & E.S. Quade. Handbook of systems analysis. Elsevier Science Publishing Co., Inc.*, New York. ISBN 0-471-92020-7.

THE EXPERIENCE OF USING A DECISION SUPPORT SYSTEM FOR NUTRIENT MANAGEMENT IN AUSTRALIA

J Richard Davis
Trevor F N Farley
William J Young
Susan M Cuddy

CSIRO Land and Water, P O Box 1666, Canberra City ACT 2601, Australia

ABSTRACT

Australia experienced an outbreak of algal blooms in rivers, estuaries and storages in the early 1990s that led to a concerted national program to reduce the eutrophication of Australian waters. Most Australian States drew up algal or nutrient management strategies which required catchment management groups to produce plans for nutrient reduction within their catchments. Australia is predominantly a rural country and diffuse nutrient sources, principally from agricultural runoff, are the major source of nutrients in most years. The catchment management groups lacked both the expertise and the tools to draw up these plans and so CSIRO, the national research organisation, produced a simple decision support system called Catchment Management Support System (CMSS) that packaged up the (then) best scientific information in an easily useable form. The program allows managers to assess the effects of Land Use and Land Management policies on nutrient loads (typically total phosphorus and total nitrogen). The policies are proposed in and the output can be presented as maps, histograms or text reports. A simple spatial model generates nutrient loads which are then transported through the drainage system using another simple routing and assimilation model. Although essentially a quantitative model, it incorporates some concepts from expert systems. For example, it can generate explanations of how it calculated its load predictions.

CMSS has now been widely adopted by managers and community groups throughout eastern Australia and has been instrumental in drawing up many nutrient management plans. Many models have been produced by scientists, very few are actually used by managers let alone community groups. The keys to the program's success lie in its design features such as its simplicity of use, very low data demand, suitability for the specific institution task and the degree of documentation, training and support offered.

KEYWORDS

Eutrophication, nutrient management, CMSS, decision support systems, Australia

INTRODUCTION

European settlers first arrived in Australia in 1788. Most settlement was confined to coastal regions until about 1820 when there was a rapid spread of Europeans into the semi-arid rangelands and tropical regions. The early development occurred in the Murray-Darling basin (Figure 1) which remains today the most important agricultural region of Australia. This development brought about major changes to the land, water and vegetation resources of the inland of Australia, including the replacement of native vegetation with pasture, the construction of reservoirs in the headwaters and weirs along the length of the rivers, and the introduction of exotic animals - principally sheep and cattle. Monitoring data show that there have been major changes in the flow patterns of the inland rivers over the last 100 years. Although water quality has been monitored over the last 20 years, it is widely believed that river water quality has deteriorated over the period of European settlement as a result of these changes. Most inland rivers are highly turbid, there are elevated levels of nitrogen and phosphorus and the rivers are increasingly saline.

There was a 1000 km long bloom of *Microcystis aeruginosa* and *Anabaena circinalis* in the Darling River of New South Wales in late 1991. The bloom was toxic. A number of animals died and drinking water supplies had to be brought in. This event received widespread publicity and captured the Australian public's attention for some weeks. The governments of New South Wales and Victoria instituted inquiries into the cyanobacterial and algal blooms and eventually developed management strategies (BGATF, 1992; SGV, 1995). The Murray-Darling Basin Commission (MDBC), an inter-governmental coordinating agency, also developed an Algal Management Strategy (MBDMC, 1994). All these strategies place considerable emphasis on nutrient reduction as a mechanism to reduce the frequency of blooms although it was recognised that this must be a long term strategy since the riverine sediments already contained a large pool of nutrients. The NSW and MDBC strategies also included the need to use flow management as another, shorter-term tool for controlling blooms.

[Fig 1 about here]

Phosphorus is the principal nutrient to be tackled in the nutrient reduction components of these strategies, partly because it is easier to manage than nitrogen and partly because phosphorus reduction will bias waterbodies away from toxic cyanobacteria. Phosphorus enters waterways from both diffuse and point sources. However, over the long term, diffuse sources dominate both phosphorus and nitrogen loads because of the low population density in inland Australia (GHD, 1992). Rainfall is highly variable and point sources can dominate in dry years.

The processes controlling the liberation and transport of diffuse source phosphorus in Australian catchments were not understood in 1992 and are only now becoming clear. Extensive gully erosion occurred in areas of duplex soils that have been cleared for agriculture over the last 100 years. Radio-isotopic tracing shows that the majority of the sediments and the particulate-bound phosphorus came from the subsoils in these areas (Wallbrink *et al.*, 1997). Fertiliser additions and the breakdown of organic matter were relatively minor contributors in these areas. In intensive irrigation districts, particularly dairying areas, fertilisers have been shown to enter rivers directly in a highly available form (Nelson *et al.*, 1996). Phosphorus also reaches the waterways in irrigation drains from irrigation districts (Harrison, 1994) and the outfalls of sewage treatment plants (STPs). Although, the GHD report shows that these point sources are relatively minor contributors to the overall phosphorus load, they may still be important since their phosphorus is presumed to be more available for uptake by pants and cyanobacteria than is particulate-bound phosphorus from diffuse sources.

The NSW Algal Management Strategy required the Total Catchment Management group in each catchment in the State to draw up a nutrient management plan. These TCM groups had been set up some years earlier by the State government to take responsibility for planning, coordinating and implementing environmental management activities in these catchments. They were a deliberate attempt to bring community groups into the process of environmental management (Cameron *et al.*, 1996). A community representative chaired each TCM group and, although State government agency staff were on each TCM, the majority of members came from local groups such as local government bodies and various special interest groups.

Whilst these TCM groups lacked the technical skills to draw up nutrient management plans themselves, they could draw upon the resources of government agencies for this assistance. However, in the spirit of TCM, the community groups would all be closely involved in the development of the plans. The Catchment Management Support System (CMSS) was the software chosen to help produce these plans.

THE CATCHMENT MANAGEMENT SUPPORT SYSTEM

CMSS is a simple decision support system developed in the early 1990s by CSIRO to help non-specialists analyse how land use and Land Management policies are likely to impact on the nutrient status of their catchment. It was originally developed for use in the Onkaparinga catchment of South Australia (Davis *et al.*, 1991b) and has now been applied to a number of catchments throughout Australia (e.g. Davis *et al.*, 1991a).

The Program has been designed around a set of principles listed below. Full details of the software design are in Davis *et al.* (1997).

The Input

1 The users should be able to propose policies directly to the program in an English-like syntax.
2 The program's data demands should be minimal.
3 The best available data and knowledge should be used by the program to predict the consequences of the policies. The sources of this information should be recorded.

The Model

4 The users should not need modelling skills to run the program.
5 There should be a balance between the precision of the policy statements, the availability of data and the complexity of the model.

The Output

6 The predictions of policy effects should be presented as simply and graphically as possible.
7 The users should be informed of the reliability of the predictions.
8 The users should be able to understand how the predictions were calculated.
9 The users should be able to see the relationship, directly, between each input (viz. each policy) and each output (viz. the predicted loads and costs).

These principles reflect the purpose that the DSS will be put to, the types of users, and the availability of data. Thus, the input to the DSS will be policies (in English) not quantitative data and the output will be maps and histograms that are immediately understandable by non-technical groups. The model that drives the DSS needs to be simple, given the paucity of data and understanding in most catchments, and these data need to include both qualitative and quantitative data. The TCM groups were basing their nutrient management plans partly on CMSS and so they needed to be convinced of its predicted nutrient loads under different policy options. Consequently, CMSS needed to be able to answer questions about its predictions and needed to provide estimates of the uncertainties in those predictions.

CMSS is structured into five major Modules (Fig 2). The TCM group describe features of the catchment and the plan in the Setup Module. For example, in CMSS, the catchment is divided into a number of subcatchments and each can be further divided into smaller units called mapping units. The coordinates of the boundaries of the subcatchments can be entered at setup time so that the program can draw maps of the predicted nutrient loads from each sub-catchment.

[Fig 2 about here]

There are four main data items held in the Data Base Module. The first of these is the area of each land uses in each mapping unit in the catchment. Both diffuse source and point source land uses can be included. The second data item is a file of nutrient generation rates for each land use. Up to 99 nutrients can be included in the modelling although, because of data availability issues, only total nitrogen (TN) and total phosphorus (TP) are normally used. These generation rates describe the expected yield of TN or TP per hectare of each land use. The nutrient generation rates are central to the operation of CMSS. The uncertainty associated with each generation rate and text descriptions of how these values were derived are included in the data file and these data are included in the calculations (see below). A Data Book has been produced to assist in estimating these generation rates (Marston *et al.*, 1994). The third data item is a file of descriptive attributes of each mapping unit. These can be defined by the users but typically include slope, rainfall, soil characteristics and vegetation cover. The fourth data item is a file describing the characteristics of the management practices to be implemented under the Nutrient Reduction Plan. The expected effectiveness of each management practice, the cost of implementing it, and the extent to which it is currently employed in the catchment are included in this file. CMSS can estimate the in-stream assimilation of nutrients. Data on time of travel of stream flow in each subcatchment and the typical nutrient assimilation rates can be included in these files, if the assimilation component is used.

The TCM groups can propose various policies in the Policy Modules. Land Use policies describe proposed changes in land use; Land Management policies describe the implementation of various practices that may improve the quality of water from either point or diffuse sources. The policies are expressed in a formal syntax and can be built up in a restricted form of English by picking items off menus. An example is given in Figure 3.

Policies are normally grouped into policy sets which describe a complete nutrient management plan. A typical plan might include policies to upgrade selected STPs, to reduce soil erosion in selected soil types within the catchment and to stabilize stream banks to reduce bank erosion and to install artificial wetlands in sensitive parts of the waterbody. A number of alternative policy sets (plans) can be proposed and stored in a library via this module.

The Model Module predicts the annual average nutrient loads at various points in the catchment. The model first estimates the base load of nutrients being generated within the catchment by simply multiplying the area of each land use ($Area_{ij}$) by its generation rate (G_i) and summing over the land uses and the spatial units.

$$Load_k = \Sigma\Sigma\, Area_{ij} * G_i \qquad (1)$$

The index i describes the land uses, j describes the spatial units and k describes the nutrients being modelled. The Model component also calculates the error on these load estimates.

This very simple nutrient generation model is appropriate for the development of nutrient management plans, given the limited data and process knowledge available, the difficulty of calibrating water quality models over large catchments and the skills of the audience that it is intended for. The effects of environmental factors such as slope, soil type, ground cover, etc are partly incorporated through the definition of the land uses. Thus if one or more of these factors is believed to be important then the 'land use' can be redefined to include it. For example, in the application of CMSS in the Murrumbidgee catchment (Cuddy *et al*, 1997) there were eight types of dryland cropping depending on the soil type and rainfall class where the cropping occurred. Thus, the effects of environmental variables are incorporated into the different generation rates for these eight land use types and are not modelled explicitly. At the time when CMSS was developed, there was insufficient knowledge about the processes to be able to produce more detailed nutrient generation models.

The nutrients generated in each subcatchment are routed through the catchment using a simple connectivity and proportioning method. In-stream assimilation due to both biotic and sedimentation processes can be represented by a simple exponential decay model, first developed in the Hawkesbury-Nepean catchment (SPCC, 1983)

$$Load_{jk}' = Load_{jk} . e^{-lt} \qquad (2)$$

where $Load_{jk}$ represents the load of nutrient k generated in subcatchment j (the prime denotes the load after assimilation). The index l is a rate constant describing the loss process and t is the average travel time through the subcatchment. The rate constant can be obtained by using an empirical relationship with the depth of the waterbody (Simmons, pers. comm.).

The members of a TCM group can test the effect of a set of policies. CMSS will firstly modify the area of land uses in each mapping unit as described in the Land Use policies and then modify the nutrient generation rates as a result of the management practices described in the Land Management policies. Then the model will be rerun to calculate the likely average annual nutrient loads after these policies have been applied.

The Interrogation Module stores the results of each calculation at mapping unit and land use level in a data base. Thus, it can construct answers to questions posed by the TCM members as they seek to gain confidence in the model's predictions. For example, it can identify which land use was the greatest contributor to the TN load in a particular subcatchment; the land use policy that had greatest effect in reducing the TP load; or the part of the catchment most affected by a certain land management policy.

APPLICATION TO THE HAWKESBURY-NEPEAN CATCHMENT

The Hawksebury-Nepean catchment covers an area of about 22,000 km^2 to the south west of Sydney (Fig 1). Land use in the catchment includes large areas of bushland (68%), much of which is national park, a significant area of unfertilised grazing (22%), and a number of smaller areas of more intensive land including areas of urban development and intensive vegetable growing. Point nutrient sources in the catchment include 36 STP's, 49 dairy sheds, and many stables and poultry sheds. The catchment also contains a number of large dams, upon which the city of Sydney depend for its water supply.

Bounded to the east by the ocean, the city of Sydney has spread both up and down the coast, and westwards into the Hawkesbury-Nepean catchment. The pressures of continued urban and peri-urban development in the catchment, as well as the increase in rural industries supplying food to the city, led to considerable concern for the water quality in the rivers and dams of the catchment. The (then) Sydney Water Board commissioned CSIRO to apply CMSS to the catchment, and in particular, as a first step in investigating a range of possible strategies for minimising nutrient loads from STP's, and controlling nutrient losses from developing and recent developed urban areas.

With no reliable land use information for the catchment, a major land use mapping exercise was undertaken. A comprehensive review of all available nutrient data was also made, and water quality data from STP monitoring in the Nepean River was used to set-up the assimilation component of the model.

Table 1 shows the base load of nutrients (current conditions) for the major contributing land uses. In CMSS these results can be viewed as a text report, map or bar chart. Loads can also be given by catchment regions (or sub-catchments) as reports, bar graphs and maps.

[Table 1 about here]

Figure 3 shows the sequential definition of a policy of land management changes. The first selection is the land use to be managed ('sewered urban'), and the second selection is the management practice ('sediment trapping pit and mini-wetland') to be used. In Figure 2 the policy is made unconditional. However, it is possible to specify condition statements (again from list menus) to allow more precise direction of where the policy is to be applied. Also, a compliance value can be specified to model partial up-take of new policies.

[Fig 3 about here]

Following a policy run, CMSS will report loads under the new conditions, and differences (both relative and absolute) in loads and in average areal export rates. These can be viewed as reports or bar charts of maps, by catchment region (or sub-catchment) or by land use. Figure 4 shows a map of phosphorus load difference (absolute) between the base and policy (before/after) runs.

[Fig 4 about here]

SUMMARY

Over 60 copies of CMSS have now been sold and it has now been applied to 10 major catchments in NSW by TCM groups. It has also been used extensively by the Queensland government, and various local groups in Victoria and New South Wales. There has been no formal survey of why this DSS has proven to be so widely adopted. However, the following reasons are mentioned in meetings and training sessions with users.

First, the DSS has the institutional backing of the NSW government. This gives it credibility with TCM groups and provides resources to help these community groups develop the data and run the model. This may be the most important reason. However, it cannot be the only reason since CMSS does not have institutional backing in other States in which it is used. Unlike, say, most water quality models, CMSS was designed for a specific purpose and user group. Thus it is based on a set of assumptions about data availability, method of communication (e.g. policies and maps) and model features (error calculations, dollar costs, explanation of results). In practice some of these features have proven very popular and others have been little used in practice. A third important reason is the level of on-going support provided. Users can contact CMSSs developers on a hot line or via email for advice. Even though the software is very easy to use, this has provided them with the confidence to use it and has not proven too onerous on the developers.

REFERENCES

BGATF (1992). *Blue-Green Algae: Final report of the New South Wales Blue-Green Algae Task Force.* Department of Water Resources, New South Wales, Australia.

Cameron, J., Macpherson D. and Gomez-Fort, R. (1996). *Integrated catchment management as communicative rationality.* Report to LWRRDC. University of Western Sydney, Richmond, NSW.

Cuddy, S.M., Young, W.J., Davis, J.R. and Farley, T.F.N (1997). Trialing the Catchment Management Support System in the Murrumbidgee catchment, New South Wales. In: *Managing Algal Blooms - Outcomes from CSIRO's Multi-Divisional Blue-Green Algal Program*, J.R. Davis (Ed.), CSIRO Land and Water, Canberra, Australia, pp 103-113.

Davis, J.R., Collins, A.G. and George, P.R. (1991a). Applying a simple catchment management support system to the Peel-Harvey catchment. *Proceedings, International Hydrology and Water Resources Symposium.* Institution of Engineers, Canberra, Australia, pp 710-713.

Davis, J.R., Nanninga, P.N., Biggins, J. and Laut, P. (1991b). Prototype decision support system for analysing impact of catchment management policies. *Journal of Water Resources Planning and Management*, 117(4), 399-414.

Davis, J.R., Farley, T.F.N. (1997). CMSS: Policy analysis software for catchment managers. *Environmental Modelling and Software*, in press.

GHD (1992). An investigation of nutrient pollution in the Murray-Darling Basin River System. Report prepared by Gutteridge Haskins and Davey for the Murray-Darling Basin Commission, Canberra, Australia. ISBN 1 875209 13 1.

Harrison, J. (1994). *Review Of Nutrients In Irrigation Drainage In The Murray-Darling Basin.* CSIRO Division of Water Resources, Water Resources Series No.11, Canberra Australia.

Marston, F.M., Young W.J. and Davis, J.R. (1993). *Nutrient Generation Databook.* Unpublished CMSS Reference Manual, CSIRO Division of Water Resources, Canberra, Australia.

Nelson, P.N., Cotsaris, E. and Oades, J.M. (1996). Nitrogen, phosphorus, and organic carbon in streams draining two grazed catchments. *Journal of Environmental Quality*, 25(6), 1221-1229.

MDBMC (1994). *The Algal Management Strategy For The Murray-Darling Basin.* Murray-Darling Basin Ministerial Council, Canberra, Australia.

SGV (1995). *Nutrient Management Strategy Of Victorian Inland Waters.* State Government of Victoria, Australia, Melbourne, Australia. ISBN 0 7306 6054 0.

SPCC (1983). *Water Quality in the Hawkesbury-Nepean River: A Study and Recommendations.* State Pollution Control Commission, Sydney, Australia.

Wallbrink, P.J., Murray, A.S., Olley, J.M. and Olive, L. (1997). Determining sources and transit times of suspended sediment within the Murrumbidgee tributaries, NSW, Australia, using fallout ^{137}Cs and ^{210}Pb. *Water Resources Research*, in press.

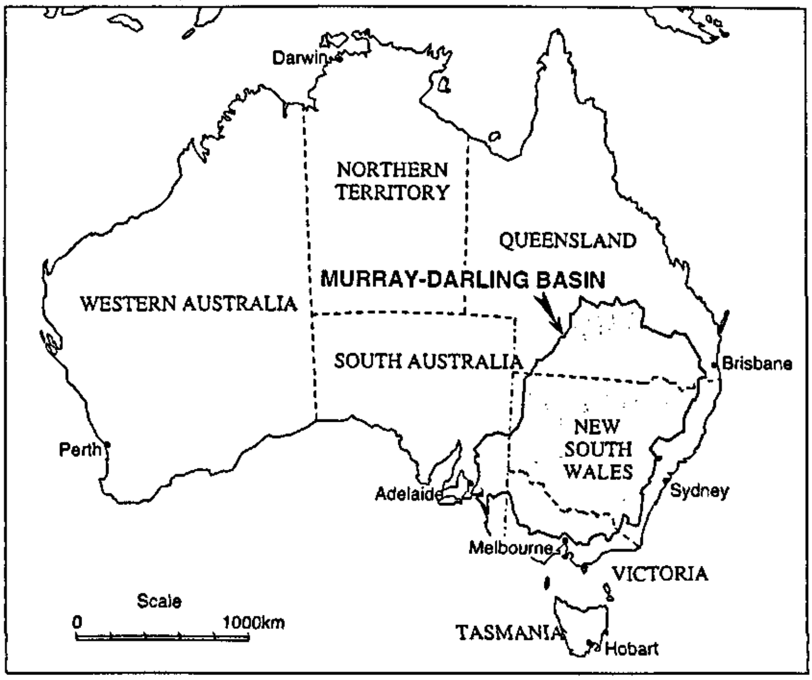


Figure 1: Australia and the Murray-Darling Basin

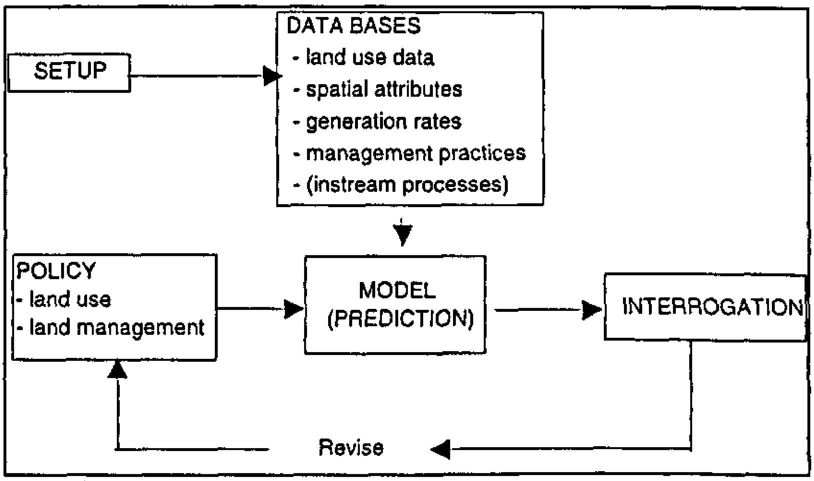


Figure 2: the five major modules of CMSS

Land Use	TP Annual average (tonnes/yr)	TN Annual average (tonnes/yr)
Bushland	148	2,218
Unfertilised grazing	119	427
Disturbed land	76	239
Fertilised grazing	65	416
Unsewered peri-urban	43	289
Recent sewered urban	29	93
Established sewered urban	26	99
Intensive vege/turf growing	20	20
STP Quakers Hill	14	73
STP St Marys	14	375
STP Lithgow	14	23
Built up - miscellaneous	10	34
TOTALS	673	4,918

Table 1: Base nutrient loads by major land use.

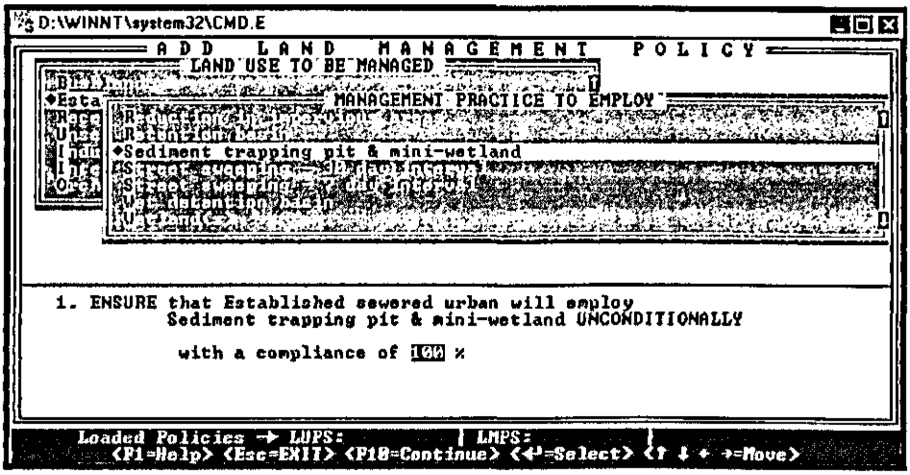


Figure 3: CMSS policy definition screen

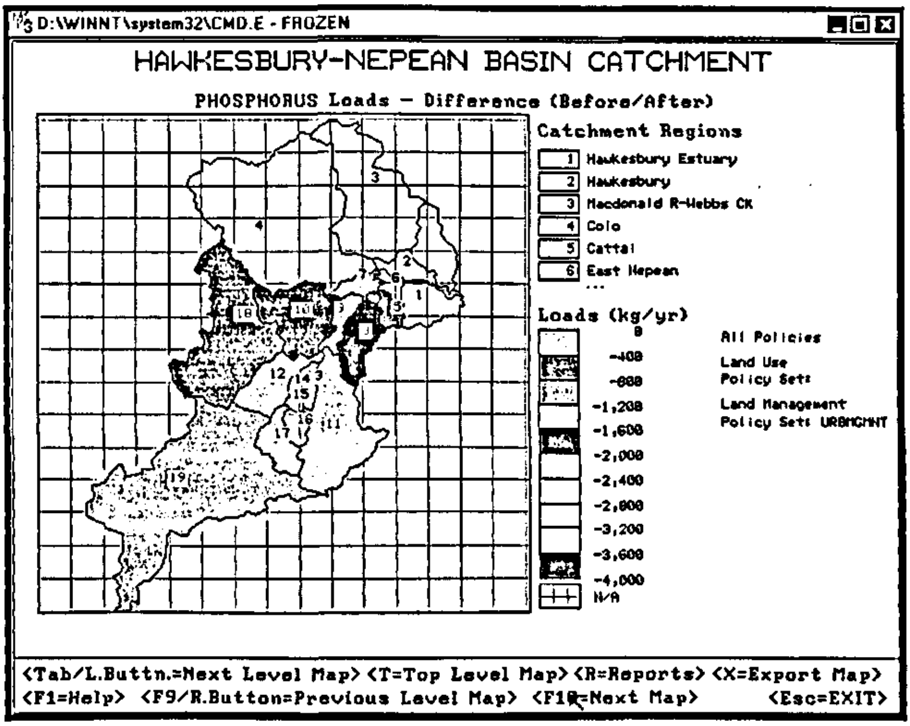


Figure 4: Catchment map showing changes in phosphorus loads following application of a Land Management policy set

ECOLOGICALLY BASED STANDARDS FOR NUTRIENTS IN STREAMS AND DITCHES IN THE NETHERLANDS

Edwin T.H.M. Peeters & Jean J.P. Gardeniers

Department of Water Quality Management and Aquatic Ecology, Agricultural University Wageningen, P.O. Box 8080, 6700 DD Wageningen, the Netherlands

ABSTRACT

For the general environmental quality legal standards for total phosphorus and total nitrogen have been formulated earlier on a nationale scale for all inland surface waters in the Netherlands. These standards are based on measurements in shallow lakes sensible to eutrophication. No legal standards are available for the specific environmental quality.
Analyses of data collected by regional waterboards in streams and ditches showed that the legal standard for the general environmental quality is not appropriate for water types other than shallow lakes. The legal standards do not provide appropriate protection for the aquatic communities. In order to protect and preserve biodiversity in ditches and streams more specific standards should be set.
Setting a single legal standard for the general enviromental quality and one for the specific environmental quality for all types of water does not hold according to the results of this study. Different types of water have different kinds of aquatic communities and different nutrient levels. It is recommended to develop water type dependent standards. In this study standards for the general and specific environmental quality for nutrients are proposed for six types of streams and for six types of ditches.

KEYWORDS

assessment; ditches; ecological; eutrophication; standards; streams; typology; water quality

INTRODUCTION

One of the central objectives of Dutch governmental policy on water quality is 'to protect and develop healthy watersystems that guarantee a sustainable use' (Anonymous, 1985). To that end two ecological objectives have been formulated (Anonymous, 1989).
The first one aims at a so-called 'general environmental quality', being a minimal acceptable water quality level for all inland waters. This basis quality level has to be reached within a limited period of time (5 years). Legal standards for the minimal water quality are set by a list of physical and chemical values and by a verbal description of the biotic community that has to be present.
The second one aims at a specific (and higher) quality level referring to natural conditions with minimized human influence. The natural conditions can differ according to type of water (Anonymous, 1988) No legal standards for this second objective are set.

With respect to nutrients the legal standards for the general environmental quality were set at 0.15 mg P/l and 2.2 mg N/l being a summer-average value for shallow waters sensible to eutrophication

(Anonymous, 1994a). The standard for phosphate is also valid as a year-average value for all other inland waters.

In recent years additional standards have been developed mainly based on the biotic components of the aquatic ecosystem. Starting from the legal verbal description (Anonymous, 1988) for each of 25 watertypes an ecological standard for the general environmental quality has been set (Klapwijk et al, 1995). Also ecological standards for higher quality levels have been proposed for each type of water. Ecological assessment systems have been made to monitor water quality and to test the new standards in the five main inland water types in the Netherlands (Anonymous, 1992; Tolkamp *et al.*,1992; Anonymous, 1993a, 1993b, 1994b, 1994c; Peeters *et al.*, 1994).

In this study the new ecological standards, specified as quality classes of the ecological assessment systems, are used to propose standards for nutrient concentrations for the minimal ecological quality and for higher quality levels in streams and ditches.

METHODS

The assessment system for streams uses the abundance of indicative macro-invertebrates to describe the ecological condition with respect to the main environmental factors (current, substratum, saprobity, eutrophication) and to the functional organization (percentage shredders, collectors, grazers). Six types of running waters are distinguished: hill streams upper, middle and lower reach and lowland streams, upper, middle and lower reach. The quality range for each environmental factor is divided into five classes, from lowest to highest quality level. To demarcate the quality classes from high to low quality the presence and abundance respectively of critical, common and tolerant indicator species is used (table 1).

Table 1. Indicator species used to demarcate quality classes.

indicator	quality level				
species	highest (5)	nearly highest (4)	middle (3)	lowest (2)	beneath lowest (1)
critical	XX*	X			
common		X	XX	X	
tolerant				X	XX

* XX: present in high numbers
X: present in low numbers

The assessment system for ditches uses the abundance of indicative macrophytes, macro-invertebrates, epiphytic diatoms and some environmental variables to describe the ecological condition with respect to the main environmental factors (eutrophication, saprobity, salinity, acidity and management). Six types of ditches are distinguished: sand-, clay-, peat-, acid, brackish and slightly brackish ditches. Like in the assessment system for streams the quality range of each environmental factor is divided into five classes and the occurrence of indicator species is used to demarcate the classes.

The middle ecological quality level of the assessment systems (class 3) conforms to the general environmental quality, the nearly highest and highest levels (class 4 and 5) to the specific environmental quality.

For this study data were used that have been collected by several waterboards from 1980 to 1990 at streams (> 4000 samples) and ditches (>1000 samples) representing the full range of quality classes. From streams only data on macro-invertebrates are available, from ditches also data on macrophytes and epiphytic diatoms.

Both from streams and ditches data on 60 environmental variables have been collected (e.g. temperature, current, pH, phosphate, nitrate, conductivity, soiltype, land-use, morphology) and on 7 management variables (e.g. function/use, maintenance measures) (Roos *et al.*, 1991). All data were stored in a database.

Data on ortho-phosphate, total-phosphate, nitrite+nitrate, ammonium nitrogen, and chlorophyll-a being primary measures of eutrophication are used to set standards for nutrients. Data on chlorophyll-a were available from ditches only. Additionally also data on conductivity, chloride and potassium are used as a check to the primary criteria: conductivity as a measure for the total ion content, chloride as a conservative tracer of the origin of water and potassium as related to agriculture.

The software packages EBEOSWA (Anonymous, 1993c) and EBEOSLO (Anonymous, 1994d) are used to calculate the ecological quality of the streams and ditches respectively.

To analyze the dataset several selections are made (table 2). In selection I all available data from the whole range of quality classes is used. Selections II, III, and IV are according to the ecological standards for the general and specific environmental quality.

Table 2. Criteria used for making selections.

selection	criteria: running waters		criteria: ditches	
I		all available data		all available data
II	current saprobic degree trophic degree	at least middle level	saprobic degree trophic degree	at least middle level
III	current saprobic degree trophic degree	at least nearly highest level	saprobic degree trophic degree	at least nearly highest level
IV	current saprobic degree trophic degree	highest level	saprobic degree trophic degree	highest level

Frequency analyses are carried out per eutrophication measure for each of the 12 types of streams and ditches using 5, 10, 25, 50, 75, 90, and 95 percentiles and mean, minimum and maximum value and number of observations.
The median values from the frequency analyses of selection IV (table 2) are used to derive the standards for the specific environmental quality. When selection IV resulted in less than 25 observations the median from selection III is used. Selection II is used to derive standards for the general environmental quality. Samples with a higher ecological quality are also present in this selection and therefore not the median value but the 75 percentile is choosen for the proposed standard.

Table 3. Results of the frequency analyses for lowland streams upper reaches.

variable selection[1]	mean	min	max	percentile 5	10	25	50	75	90	95	n[2]
orthophosphate (mg/l)											
I	0.72	0.01	17.16	0.03	0.03	0.06	0.10	0.25	0.91	3.53	329
II	0.25	0.01	7.80	0.03	0.03	0.05	0.09	0.15	0.43	0.70	160
III	0.09	0.01	0.55	0.02	0.03	0.03	0.06	0.09	0.23	0.42	33
IV	0.11	0.01	0.37	-[3]	-	0.03	0.06	0.17	-	-	9
totalphosphate (mg/l)											
I	0.99	0.02	18.00	0.05	0.07	0.13	0.25	0.66	2.08	3.96	353
II	0.41	0.02	8.50	0.04	0.06	0.10	0.18	0.40	0.77	1.33	155
III	0.20	0.02	0.77	0.03	0.06	0.09	0.15	0.22	0.52	0.75	40
IV	0.23	0.05	0.77	-	0.05	0.09	0.17	0.27	0.74	-	10
nitrite+nitrate (mg/l)											
I	4.88	0.01	31.18	0.12	0.31	0.89	2.55	7.27	13.10	17.00	396
II	5.87	0.06	31.08	0.15	0.38	1.00	3.45	9.54	15.11	18.35	176
III	4.80	0.06	31.08	0.11	0.12	0.70	2.40	7.71	13.09	20.12	39
IV	5.30	0.41	31.08	-	0.47	0.74	1.73	7.72	26.44	-	11
ammonium (mg/l)											
I	1.26	0.01	57.00	0.02	0.08	0.14	0.40	1.00	2.31	4.91	377
II	0.53	0.01	4.90	0.01	0.03	0.10	0.28	0.70	1.50	2.10	183
III	0.24	0.01	1.10	0.01	0.02	0.10	0.14	0.26	0.78	1.00	40
IV	0.18	0.01	0.87	-	0.1	0.06	0.10	0.15	0.75	-	11
conductivity (μS/cm)											
I	472	88	1287	158	193	310	473	615	737	861	329
II	432	88	1117	143	160	254	432	578	715	795	164
III	405	94	1117	95	109	175	319	616	835	954	35
IV	478	154	1117	-	-	177	325	843	-	-	9
chloride (mg/l)											
I	44	8	251	14	17	24	38	51	77	99	377
II	39	8	251	10	15	21	35	46	71	90	183
III	39	8	251	9	10	16	33	47	72	86	40
IV	59	12	251	-	12	18	31	78	218	-	11
potassium (mg/l)											
I	7.8	0.2	39.1	0.9	1.1	2.0	4.8	9.2	22.1	25.1	98
II	8.9	0.2	39.1	0.8	1.0	2.0	5.1	16.5	23.0	26.7	57
III	9.1	0.2	26	-	0.8	1.0	3.9	18.5	23.3	-	18
IV	8.0	1.1	20.0	-	-	1.1	3.0	-	-	-	3

[1]: see table 2 for selection criteria
[2]: number of observations
[3]: too few observations

RESULTS AND DISCUSSION

The results of the frequency analyses are given in detail for one type of water, lowland streams upper reaches (table 3). The overall pattern for the other 11 types of water is similar. Table 3 shows that with decreasing numbers of influenced sites (selection I to IV) the concentrations become lower, e.g. 50 percentile for ammonium decreases from 0.40 via 0.28 and 0.14 to 0.10 mg/l. In general, all calculated percentiles and minimum, maximum and average value show a decrease in concentrations with increasing constraint on quality.

From the table it is also clear that in some cases in selection IV and III only a few observations are left, e.g. potassium. Especially for three types of ditches (acid, slightly brackish and brackish) selections III and IV have too few observations. Standards for the specific environmental quality, therefore, can not be derived for these types.

Table 3 shows that the average value is in general higher than the median one, for example nitrite+nitrate in selection II gives an average of 5.87 mg/l and a median of 3.45 mg/l. The difference between mean and median value indicates there are some extreme (high) values. The differences between the average and the median are becoming smaller from selection I to selection IV. For example, the difference between mean and median for total phosphorus in selection I is 0.72 mg/l and in selection III 0.11 mg/l.

The results for conductivity, chloride, and potassium show that values become lower with increasing constraint on quality. This confirms that by selecting the higher quality levels the influenced sites are removed.

The proposed standards for the general and specific environmental quality are given in table 4. This table shows considerable differences between the standards of the 12 types of water. Nitrite+nitrate concetrations for the specific environmental quality for example ranges from 2.40 to 5.00 mg/l for streams and from 0.11 to 0.89 mg/l for ditches. Differences are found both for the general environmental quality and the specific environmental quality. From these differences it can be concluded that there is a need for water type dependent standards. In his cenotype approach Verdonschot (1990) also pointed at differences between aquatic communities and environmental variables from different types of waterbodies.

The legal standard for total phosphate (0.15 mg P/l) in the Netherlands for the general environmental quality is based on summer average values of monthly samples from shallow stagnant waters sensible to eutrophication (Anonymous, 1994a). For all other inland waters this standard is used as a year average of monthly samples. The proposed standards for the general environmental quality as given in table 4 differ from this legal standard. Only the value for acid ditches is lower than the legal standard, for all other water types the value is (much) higher. According to this study the extrapolation of the standard from shallow stagnant waters to other types of water does not hold.

Natural background concentrations of phosphate have been studied earlier in the Netherlands (Anonymous, 1988). As an upper limit for orthophosphate the value of 0.10 mg P/l is mentioned for streams and fresh water ditches and 0.20 mg P/l for brackish ditches. The values from this study for the specific environmental quality are lower for all ditch types and for upper reaches of both lowland and hill streams. For the middle and lower reaches the values are higher.

The legal standard for nitrogen is based on year average values of monthly samples from total nitrogen (sum Kjeldahl-N and nitrite+nitrate N). The values for nitrite+nitrate found in this study are higher than the legal standard for total nitrogen. The proposed standards for nitrite+nitrate are a factor 2 to 5 higher than the legal standard for total nitrogen.

Natural background concentrations for nitrate have been mentioned as being between 0 and 1 mg/l for both streams and ditches (Anonymous, 1988). The proposed standards for all types of streams are much higher.

The legal standard for the general enviromental quality for chlorophyll-a is 100 mg/l (Anonymous, 1994a). The proposed standards for ditches from this study are much lower.

Table 4. Proposed standards for nutrients for the general and specific environmental quality.

	orthophosphate (mg/l)		totalphosphate (mg/l)		nitrite+nitrate (mg/l)		ammonium (mg/l)		chlorophyll-a (mg/l)	
	GEQ[1]	SEQ[2]	GEQ	SEQ	GEQ	SEQ	GEQ	SEQ	GEQ	SEQ
hill stream upper reach	0.14	0.08	0.38	0.24	11.00	4.95	0.30	0.20	x[3]	x
hill stream middle reach	0.94	0.54	1.03	0.72	8.10	4.24	1.05	0.30	x	x
hill stream lower reach	1.00	0.80	1.35	1.00	5.00	4.65	1.59	1.30	x	x
lowland stream upper reach	0.15	0.06	0.40	0.15	9.54	2.40	0.70	0.14	x	x
lowland stream middle reach	0.36	0.14	0.76	0.18	6.51	5.64	1.20	0.37	x	x
lowland stream lower reach	0.43	0.19	0.76	0.36	6.10	5.00	2.30	0.70	x	x
sandy bottom ditch	0.06	0.05	0.32	0.08	2.12	0.34	0.65	0.27	41.0	6.5
clayish bottom ditch	0.38	0.05	0.66	0.17	2.20	0.89	0.37	0.16	39.0	10.0
peaty bottom ditch	0.07	0.05	0.28	0.14	0.34	0.11	0.40	0.20	52.0	10.0
acid ditch	0.05	-[4]	0.05	-	0.14	-	0.05	-	-	-
brackish ditch	0.20	-	0.42	-	2.30	-	1.80	-	-	-
slightly brackish ditch	0.82	-	1.90	-	1.28	-	3.40	-	104	-

[1]: general environmental quality
[2]: specific environmental quality
[3]: no data available
[4]: too few observations

CONCLUSIONS

Setting a single legal standard for the general enviromental quality and one for the specific environmental quality for all types of water does not hold according to the results of the ecological approach in this study. Different types of water have different kinds of aquatic communities and different nutrient levels. It is recommended to develop water type dependent standards.

The legal standard for the general environmental quality is not appropriate for water types other than shallow lakes. The legal standards do not provide appropriate protection for the aquatic communities.

In order to protect and preserve biodiversity in ditches and streams more specific standards should be set. In this study standards based on an ecological approach are proposed.

REFERENCES

Anonymous, (1985). *Indicative Programme Water 1985-1989.* Dutch Ministry of Transport and Water Management and Dutch Ministry of Housing, Physical Planning and Environment, The Hague, The Netherlands. (in Dutch).

Anonymous, (1988). *Ecological standard objectives for dutch surface waters.* CUWVO (Dutch Commission for Enforcement of the Water Pollution Act), The Hague, The Netherlands. (in Dutch).

Anonymous, (1989). *Third report on water management.* Dutch Ministry of Transport and Water Management, The Hague, The Netherlands. (in Dutch).

Anonymous, (1992). *Ecological assessment and management of surface waters. Assessment system for running waters based on macroinvertebrates.* Dutch Foundation for Applied Water Research, report no. 92-07, Utrecht. (in Dutch).

Anonymous, (1993a). *Ecological assessment and management of surface waters. Assessment system for ditches based on macrophytes, macroinvertebrates and epiphytic diatoms.* Dutch Foundation for Applied Water Research, report no. 93-14, Utrecht. (in Dutch).

Anonymous, (1993b). *Ecological assessment and management of surface waters. Assessment system for shallow lakes based on macrophytes and phytoplankton.* Dutch Foundation for Applied Water Research, report no. 93-16, Utrecht. (in Dutch).

Anonymous, (1993c). *EBEOSWA: Software program for the ecological assessment of running waters based on macroinvertebrates.* Dutch Foundation for Applied Water Research, Utrecht. (in Dutch).

Anonymous, (1994a). *Evaluation Report Water.* Dutch Ministry of Transport and Water Management, The Hague, The Netherlands. (in Dutch).

Anonymous, (1994b). *Ecological assessment and management of surface waters. Assessment system for canals based on macrophytes, macroinvertebrates, epiphytic diatoms and phytoplankton.* Dutch Foundation for Applied Water Research, report no. 94-01, Utrecht. (in Dutch).

Anonymous, (1994c). *Ecological assessment and management of surface waters. Assessment system for sand, clay and gravel pits based on phytoplankton, zooplankton, macrophytes and epiphytic diatoms.* Dutch Foundation for Applied Water Research, report no. 94-18, Utrecht. (in Dutch).

Anonymous, (1994d). *EBEOSLO: Software program for the ecological assessment of ditches based on macrophytes, macroinvertebrates and epiphytic diatoms.* Dutch Foundation for Applied Water Research, Utrecht. (in Dutch).

Klapwijk, S.P., Gardeniers, J.J.P., Peeters E.T.H.M. and Roos, C. (1995). Ecological assessment of water systems. In: *Proceedings Monitoring Tailor-made*, M. Adriaanse, J. van de Kraats, P.G. Stoks and R.C. Ward (Eds.). The Netherlands Institute for Inland Water Management and Waste Water Treatment, pp 105-117.

Peeters, E.T.H.M., Gardeniers, J.J.P. and Tolkamp, H.H.(1994). New methods to assess the ecological status of surface waters in The Netherlands. Part 1: Running Waters. *Verh. Internat. Verein. Limnol.* **25**, 1914-1916.

Roos, C., Gardeniers J.J.P., Roijackers R.M.M. and Peeters E.T.H.M.(1991). Ecological assessment of Dutch inland waters: Philosophy and preliminary results. *Verh. Internat. Verein. Limnol.* **24**, 2104-2106.

Tolkamp, H.H., Gardeniers, J.J.P. and Peeters, E.T.H.M. (1992). Entwicklung der ökologischen Gütebeurteilung aus der biologischen Gewässergütebeurteilung in den Niederlanden. In: *Ökologische Bewertung von Fliessgewässern*, G. Friedrich and J. Lacombe (Eds.), Band 3, Gustav Fisher Verlag, Stuttgart, New York., pp 79-95.

Verdonschot, P.F.M. (1990). *Ecological characterization of surface waters in the province of Overijssel (the Netherlands).* Ph.D.Thesis, Agricultural University Wageningen. Province of Overijssel, Research Institute for Nature Management.

Trend-analysis of eutrophication in lakes in The Netherlands

Rob Portielje and Diederik T. Van der Molen

Institute for Inland Water Management and Waste Water Treatment (RIZA), PO Box 17, 8200 AA Lelystad, The Netherlands.

ABSTRACT

A trend-analysis of eutrophication variables was performed for a large number of lakes in The Netherlands. Data of in total 232 lakes were available. Data on chlorophyll-a, total phosphorus (total-P) and total nitrogen (total-N) were analysed over the period 1980 - 1996. Summer-averaged concentrations for chlorophyll-a, total-P and total-N decreased in 65%, 73% and 75% respectively, of those 164 lakes with at least eight years data. Results for winter means are comparable (a negative trend in 54%, 77% and 69% for chlorophyll-a, total-P and total-N respectively). Since 1980 the average decrease in summer mean concentrations of chlorophyll-a, total-P and total-N is 2.61 $\mu g.l^{-1}.j^{-1}$, 0.008 $mg.l^{-1}.j^{-1}$ and 0.046 $mg.l^{-1}.j^{-1}$, respectively, illustrating the effects of (inter)national and regional measures to combat eutrophication.

KEYWORDS

Chlorophyll-a; eutrophication; lakes; total-nitrogen; total-phosphorus; trend-analysis

INTRODUCTION

Eutrophication has been a major water quality problem in Dutch surface waters over the past decades (Golterman, 1970; CUWVO, 1976; CUWVO, 1980; Lijklema *et al.*, 1988). Therefore, since the seventies measures were taken to combat eutrophication. In first instance, point sources of phosphorus were reduced in regional projects. In the eighties a national policy to reduce phosphorus emissions became effective (phosphorus removal at treatment plants, replacement of phosphorus in detergents, international agreement on reducing point sources to the main rivers). In the nineties a start was made in reducing diffuse emission from agriculture and additional measures (e.g. food web manipulation, dredging and inactivation of sediment phosphorus) were carried out in regional projects (Hosper, 1984; Van Liere and Gulati, 1992; Boers and Van der Molen, 1993; Meijer *et al.*, 1994). Several lakes recovered remarkably well from severe eutrophication. The objective of this paper is to analyse trends in eutrophication variables in lakes in The Netherlands. Data from a majority of Dutch lakes were used. A detailed description of the data, the methods and the results is presented by Portielje and Van der Molen (1997).

METHODS

Data on chlorophyll-a, total-P and total-N were collected from 1980 onwards. Before 1980, the number of lakes with suitable data was limited. Data were averaged over the summer half year (1 April - 1 October) and the winter period (1 October - 1 April). Half-year averages were calculated after linear interpolation of the measurements to individual days, to correct for non-equidistant sampling frequency. To reduce the effect of measurement errors, only those half-year periods were used from which at least five measurements were available. The lakes are represented by one sampling location. Most Dutch lakes are shallow (< 6 m) and the surface area of the lakes studied varies between a few hectare and more than 100 square kilometres.

The Mann-Kendall test was applied to quantify the trends for the individual lakes. In this non-parametric test the trend is estimated by the median of the set of the slopes (Theil slope estimator, Theil, 1950; KIWA, 1994) that are calculated from all possible pair wise combinations of the data (in this study the summer or winter means). This method is more robust than linear regression. The distribution of the residuals (differences between measured and calculated value) is usually not uniform and linear regression is more sensitive to extreme values. For individual lakes, we only used those variables for trend analysis for which summer or winter means from at least eight years were available.

Apart from trends for individual lakes, a general trend in eutrophication variables for all Dutch lakes over the period 1980 - 1996 was determined from the annual median of the distribution of the summer mean concentrations of all 232 lakes. Finally, to estimate the effect of missing years in this total set of lakes, these trends were also analysed for a smaller subset of lakes from which data were available for almost every year since 1984.

RESULTS

In a majority of the lakes a decrease in summer mean chlorophyll-a, total-P and total-N concentrations can be observed (Table 1). In many cases this trend is significant at $p<0.1$. In the majority of the lakes (67 %) the N/P ratio increased, indicating that the reduction in total-P is higher relative to the reduction in total-N. Trends in summer and winter are comparable, although the relative decrease in the winter mean chlorophyll-a was smaller than that of the summer means. The relative decrease in total-N on the other hand was larger with respect to the winter mean. The number of lakes with sufficient data to calculate means is lower in winter compared to the summer.

Table 1. Eutrophication trends in individual lakes. Significance (at $p<0.1$) is expressed as the percentage of the total number of lakes for each variable.

	Summer				Winter			
	number of lakes	negative median trend	significant negative	significant positive	number of lakes	negative median trend	significant negative	significant positive
Chl-a	160	66 %	44 %	11 %	106	54 %	37 %	11 %
Total-P	164	73 %	48 %	12 %	120	77 %	53 %	12 %
Total-N	140	75 %	39 %	13 %	104	69 %	42 %	13 %

Figure 1 illustrates the medians of the distributions of summer mean concentration of chlorophyll-a, total-P and total-N and their 10% and 90% percentiles over 1980 - 1996. Trend-analysis on the median concentrations of the half-year means shows a significant decrease for all three variables for both summer and winter (Table 2). The available data did not cover the whole period 1980-1996 for each lake. Therefore the set of lakes that makes up the annual distributions in Figure 1 varies from year to year. To check whether this variation may have affected the results, a trend-analysis was performed on a smaller fixed set of lakes with a long time-series of measurements. As for many lakes data were missing from the period 1980-1983, this analysis was performed for the period 1984-1996 with those lakes for which not more than one year of data in this period was missing. The trends for summer mean chlorophyll-a, total-P and total-N were now -4.24 $\mu g.l^{-1}.j^{-1}$ (n=89), -0.009 $mg.l^{-1}.j^{-1}$ (n=100) and -0.036 $mg.l^{-1}.j^{-1}$ (n=72) respectively. Consistent with the results of trend-analysis on the total data set, for this subset also negative trends in concentrations were observed.

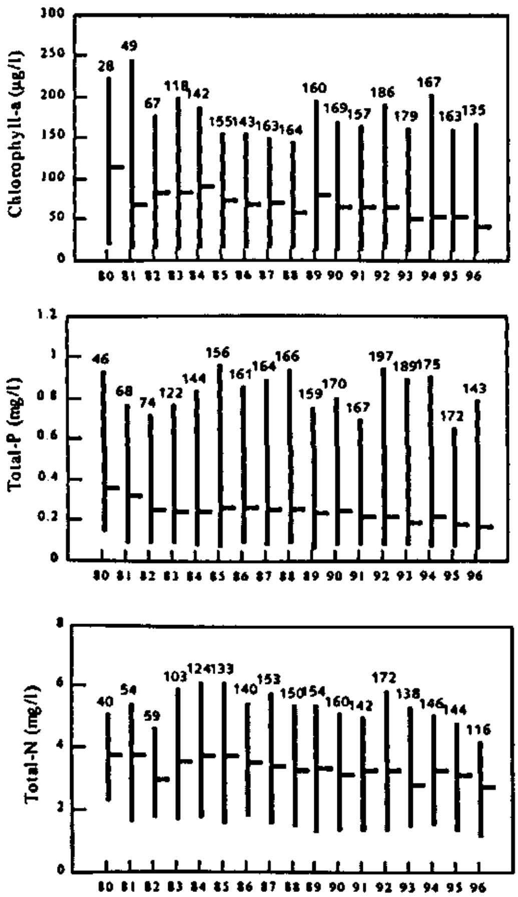


Figure 1 Median summer mean concentrations for chlorophyll-a (top), total-P (middle) and total-N (bottom), and the 10% and 90% percentiles of their distributions during the period 1980 -1996. Labels indicate the number of lakes that make up the distributions.

Table 2. Absolute trends and 10% and 90% percentiles and relative trends ($\%.j^{-1}$) in the annual median of summer and winter mean concentrations in all lakes (1980 -1996).

		Summer				Winter			
	unit	10%	median	90%	relative trend	10%	median	90%	relative trend
Chl-a	$\mu g.l^{-1}.j^{-1}$	-3.65	-2.61	-1.98	-3.6	-1.39	-0.84	-0.22	-1.8
Total-P	$mg.l^{-1}.j^{-1}$	-0.010	-0.008	-0.006	-3.3	-0.009	-0.007	-0.006	-3.0
Total-N	$mg.l^{-1}.j^{-1}$	-0.060	-0.046	-0.034	-1.3	-0.105	-0.082	-0.058	-2.3

DISCUSSION

Data of a large number of lakes were used to analyse trends in eutrophication variables of lakes in The Netherlands. Trends were analysed for means over summer and winter half year. When considering a period within an annual cycle that is too short, the trend-analysis is sensitive to shifts in the biological cycle from year to year. If this period is chosen too long, then trends may not be detected due to considerable variations of controlling variables within that period. For example, in many lakes the hydrology in winter is determined by surplus water from surrounding areas (polders), while in summer the lake may be dominated by water from other areas, e.g. imported by the river Rhine. Furthermore, using summer mean concentrations is consistent with previous studies in The Netherlands and elsewhere. The results of trend analysis for winter and summer periods are similar.

The fraction of the lakes with a negative trend was slightly higher with respect to nutrients than for chlorophyll-a. Ratio's of chlorophyll-a to total-P and total-N are to a certain extent coupled to each other by the intracellular content of the phytoplankton. However, nutrients limit phytoplankton growth only below a certain nutrient level. In fact, the percentage of lakes with a decreased chlorophyll-a concentration is surprisingly large considering the ranges in nutrient concentrations. Also other factors, such as top down control, may have affected chlorophyll-a. Furthermore, the decrease in chlorophyll-a will be non-linear with nutrient concentration in the range where nutrients become limiting, as the ratio's of chlorophyll-a : P and chlorophyll-a : N inside of the cells may decrease with decreasing external concentrations due to adaptation of the algae.

A comparable number of lakes showed a decrease in the total-P and total-N concentration. The N/P ratio generally increased, which is due to a higher relative decrease in the total-P concentration compared to total-N. Most nutrient emission measures have been focused on phosphorus. Between 1985 and 1995 phosphorus emissions have been reduced by more than 50% (Van der Molen *et al.*, 1997), while nitrogen emission was reduced less than 25%. On the other hand, it is known that the response time of phosphorus to a decreased external loading is relatively long (Jeppesen *et al.*, 1991; Van der Molen and Boers, 1994). However, the time-scale of this delay is in the same order of magnitude as the period used for trend-analysis in this study. The negative trend in nitrogen may also be induced by the negative trend in phosphorus. As the amount of nutrients built into organic matter is reduced due to phosphorus limitation, a larger fraction of the nitrogen loading occurring as dissolved components will remain dissolved. This fraction is more susceptible for removal by denitrification of either allochtonous nitrate or nitrate formed in the system by nitrification

Besides for individual lakes, trend-analysis was also performed on annual distributions of summer mean concentrations in all lakes over the period 1980-1996, to determine a national trend. However, the set of lakes that make up these distributions varies from year to year. The number of lakes with available data increased from 1980 onwards. Probably, in the earlier years only the most important (e.g. largest) water bodies have been monitored, while later on also smaller lakes are added to monitoring programs. However, the results were comparable for both approaches.

CONCLUSIONS

- Most lakes in The Netherlands show a decreasing tendency in the period 1980 - 1996 for chlorophyll-a, total-phosphorus and total-nitrogen. Therefore, it can be concluded that measures to combat eutrophication have been effective. Nevertheless, in a small fraction (≈11-13%) of the lakes the nutrient and chlorophyll-a concentrations were still significantly ($p<0.1$) increasing. These lakes deserve further attention.
- The relative decrease in total-P is larger than the decrease in the total-N. The relative decrease in chlorophyll-a is high compared to the ranges and the decreases in nutrient concentrations, suggesting that other factors are also important to control phytoplankton growth.
- Besides concentrations in individual lakes, the annual median concentrations also decreased on a national scale since 1980. This conclusion is valid for both the total set of 232 lakes, as well as for the fixed subset of lakes with long time-series of data.
- Trends in the summer and winter half year were comparable, despite possible differences in hydrology and biological factors.

ACKNOWLEDGEMENT

This study was financed by the Institute for Inland Water Management and Waste Water Treatment (RIZA), Lelystad, The Netherlands, and the Foundation for Applied Water Research (STOWA), Utrecht, The Netherlands.
We gratefully acknowledge the support of the Dutch0 regional water managers. It would not have been possible to carry out this research without the large amount of data they supplied.

REFERENCES

Boers, P.C.M. and Van der Molen, D.T. (1993). Eutrophication control in Europe - the state of the art. *Europ. Water Poll. Contr.*, 3, 19 - 25.

CUWVO (1976). Schets van de eutrofiëringssituatie van het Nederlandse oppervlaktewater en overzicht van de onderzoeksactiviteiten van de verschillende waterkwaliteits-beheerders. Resultaten van de eerste eutrofiëringsenquête. Coördinatie commissie Uitvoering Wet Verontreiniging Oppervlaktewateren, RIZA Lelystad. (in Dutch)

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

Golterman, H.L. (1970). Mogelijke gevolgen van fosfaateutrofiëring van het

oppervlaktewater. *H2O* **3**, 209-215. (in Dutch)
Hosper, S.H. (1984). Restoration of Lake Veluwe, The Netherlands, by reduction of phosphorus loading and flushing. *Wat. Sci. Tech.*, **17**, 757-768.
Jeppesen, E., Kristensen P., Jensen J.P., Sondergaard M., Mortensen E. and Lauridsen, T. (1991). Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. *Mem. Ist. Ital. Idrobiol.*, **48**, 127-148.
KIWA (1994). A protocol for trend analysis of the input of pollutants to the North Sea. SWO 93.337.
Lijklema, L., Janse, J.H., Roijackers, R.M.M. and Meijer, M-L. (1988). Eutrofiëring in Nederland. *H2O,* **21**, 462-467.
Meijer, M.L., Jeppesen, E., Van Donk, E., Moss, B. Scheffer, M., Lammens, E.H.R.R., Van Nes, E.H., Van Berkum, J.A., De Jong, G.J., Faafeng, B.A. and Jensen, J.P. (1994). Long-term responses to fish-stoch reduction in small shallow lakes: interpretation of five-year results of four biomanipulation cases in The Netherlands and Danmark. *Hydrobiologia,* **275/276**, 457-466.
Portielje, R. and Van der Molen, D.T. (1997). Trendanalyse eutrofieringstoestand van de Nederlandse meren en plassen. Deelrapport I voor de Vierde Eutrofiëringsenquête. RIZA nota. (in Dutch)
Theil, H. (1950) A rank-invariant method of linear and polynomial regression analysis, 1,2 and 3. *Ned. Akad. Wetensch. Proc.*, **53**, 386-392, 521-525, 1397-1412.
Van der Molen, D.T. and Boers, P.C.M. (1994). Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading. Proceedings International Conference on Nutrient Dynamics and Biological Structure in Shallow Freshwater and Brackish Lakes. 16-20 August 1992, Silkeborg, Denmark. *Hydrobiologia,* **275/276**, 379 - 389.
Van der Molen, D.T., Breeuwsma, A. and Boers, P.C.M. (1997). Agricultural nutrient losses to surface waters in the Netherlands: impact, strategies and perspectives. *J. of Environ. Qual.*. In press.
Van Liere, L. and Gulati, R.D. (1992). Restoration and recovery of shallow eutrophic lake ecosystems in The Netherlands: epilogue. *Hydrobiologia*, **233**, 283-287.

PUBLIC ADMINISTRATIONS: THE MAIN CAUSE OF THE TRASIMENO LAKE EUTROPHICATION

Linda Cingolani, Elisabetta Ciccarelli,
Margherita Di Brizio, Giuliana Taramella

Dipartimento di Prevenzione, Laboratorio Biologico, ULSS N. 2, V. XIV Settembre, 75. 06100Perugia, Italy

ABSTRACT

Investigations on Trasimeno lake pointed out the immission of heavy organic loads, coming from creeks, ditches and wastewater treatment plants. The consequences were progressive eutrophication phenomena: summer algal blooms (*Aphanizomenon flos aqae*, *Anabaenopsis circularis*, *Anabaena* sp, *Phormidium* sp, toxin producers), mosquito and fly invasions, changes in the fish community composition, unpleasant odours of the drinking water drawn from the lake. The risks for human population living in the area may regards not only economic damages and landscape unactractive evolution, but propagation and increase of tumor promoter phytotoxins. Administrations ought to intervene with appropiate actions to save lake and safeguard population. Ignorance, inefficiency, dubious businesses and incommunicability, instead, had promoted, until now, only propagandistic campaigns on the good condition of the lake and the adoption of mild remedies as the use of yellow panels to attract mosquitos. Only after many complaints against municipal administration, at the end of summer 1996 active carbon filters was added, but no reclamation action was made to improve lake conditions.

KEY WORDS

Administration inefficiency; Algae; *Anabenopsis circularis*; *Aphanizomenon flos aqae*; eutrophication; phytotoxins.

INTRODUCTION

The present work means to show how ignorance, inefficiency, dubious businesses and incommunicability are the main causes of environmental damages.

In Italy, surely, the water cycle culture is unknown . The water is cut to bits, and each of these parts is administered independently: drinking water, surface water, waste water, groundwater.

The law limit for some chemical substances is, often, the only one criterium for evaluating water quality, ignoring the intricate relationships that the water flows entertain with lands and human activities in the whole watershed.

In Umbria, central region of Italy, a progressive deterioration is striking the Trasimeno lake, utilized, among other things, for bathing, fishing and drinking water production. The eutrophication problems of the lake, the largest in central Italy, comes largely from the inadequate behaviour of both municipal, districtual, regional administrators and local sanitary authority. Repeated warnings of our Environmental Biology and Toxicology Control Service on the excessive amount of wastes from urban areas and factories discharged into the lake were always unheard. The authorities did not consider these factors important for maintaining the good quality of the aquatic resource. The evolution of ecological parameters was completely neglected in the years, while the immission of organic loads was favoured through treatment plants discharging in the lake (Cingolani & Ciccarelli, 1993; Cingolani, in press) and the propagation of factories in the watershed. The increase of Algal blooms, mosquito invasions, drinking water stinks, changes in fish populations, water torbidity, pH and dissolved oxygen are considered natural phenomena of a natural aquatic environment. The aim of the paper is to point out how the routinary surveillance program, unitely to proper observations on the area features, may be sufficient to give appropriate informations. From some years, instead, public administrations, employing many pecuniary resources, are committing studies to local research institutes and private firms, which, sometimes, show a tendency to sell twice their works. The results of such researches have never given a final answer on the lake conditions, overall as regards solutions to adopt (except a project regarding the connection of other creeks to increase the water level), allowing the local sanitary units to ignore our data and remarks. We will not assert that studies are useless, but that researches might be entrusted to really expertise consultants, convolving control structures, in order to have solutions to problems.

THE STUDY AREA

The Trasimeno lake comes from tectonic and alluvial phenomena, having a water surface of 120 Kmq, a perimeter of 54 km, a medium depth of 4,7 m, a max depth of 6,3 m, water turnover time of 25 years. Small streams and ditches feed the lake, some of which dry in summer. From about ten years, the increasing water catchment and rainfall decrease have reduced water thickness under the level of the spillway. The lake, therefore, cannot change its water and is concentrating nutrients and other external substances. About 60.000 i.e. gravitate on the Trasimeno area, doubling in the summer in consequence of high tourist flows. Seven purification plants, now reduced to 5, were built on the shores of the lake, to treat civil wastewaters. The effluents are discharging into the lake. In two maps worded by the regional "Office of Urbanistic-Territorial Plan" one may observe how a large part of the area contains many factories and is destined to fertirrigation (REGIONE UMBRIA, 1995), while another map underlines that the same area is considered particularly interesting from an agricoltural point of wiew. The map of the factories disseminated over the watershed shows clearly the strength of the polluting pressure of these activities Fig. 1.

Fig. 1 - Factories around Trasimeno lake (REGIONE UMBRIA P.U.T., 1995).
(Scale of 1: 200.000)

Contemporary, the Urbanistic-Territorial Plan declares that the stretch of water is Regional Lacual Park, while the lands around the lake are considered of great naturalistic quality, with European Community proposals for special protection programs (Fig. 2).

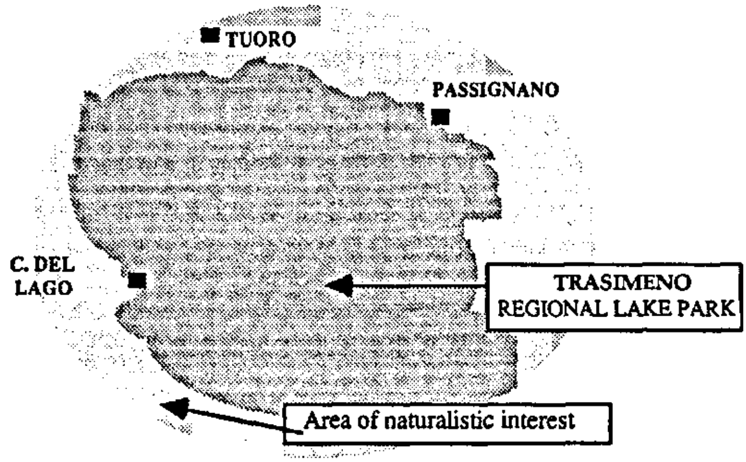


Fig. 2 - Localization of the lake park and area of naturalistic interest around the Trasimeno lake. (REGIONE UMBRIA P.U.T., 1995).

MATERIALS AND METHODS

The lake conditions are monitored every year by our Environmental Protection Laboratory in conformity with the actual italian lows for bathing, water catchment for drinking and eutrofication. The Biology and Toxicology Control Service, in particular, must control chlorophyl-a (DM 16.6.88; I.R.S.A., 1995) and algae (BOURELLY, 1985) in 7 sampling points for euthrophycation problems; faecal indicator microorganism presence (IRSA, 1995) is controlled in 14 sampling points for bathing. In the present work we discuss the data of chlorophyl and the behaviour of algal populations from 1992 to 1996. Samples have been collected at the 7 stations once a month from October to March, twice a month from April to September. In summer 1996 an investigation was also carried out on faecal microorganism presence in 25 tributaries, for a total of 32 samples. Macroinvertebrate communities of the largest creeks were also observed (Ghetti, 1995) in order to evalue the quality of the water put in the lake. Three series of samples, in winter, summer and autumn, were collected from the 5 treatment plant effluents to measure the faecal microbiological load.

RESULTS AND DISCUSSION

The conditions of the tributaries, as regards faecal pollution, is showed in Fig. 3-4.

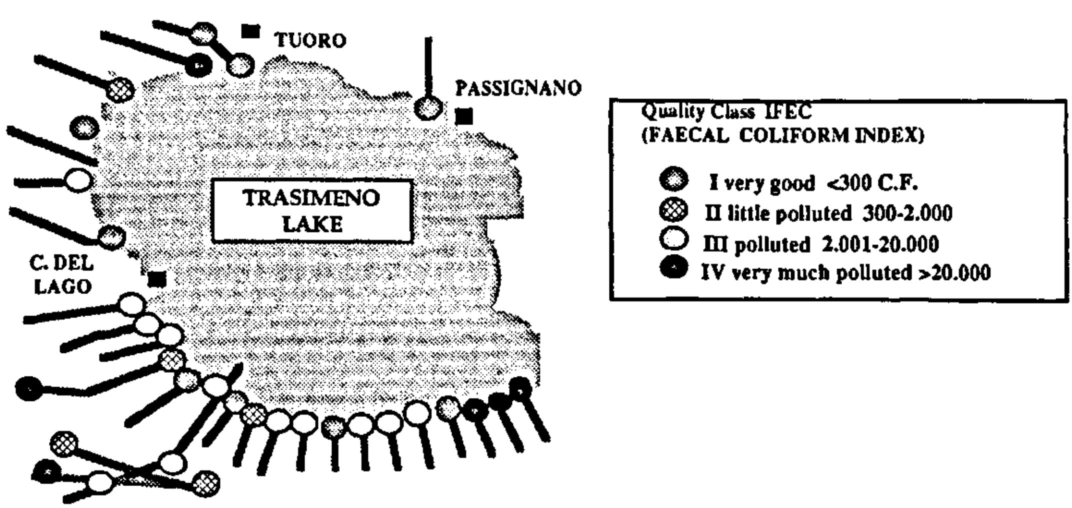


Fig. 3 - Faecal contamination of the lake tributaries registered in 1996.

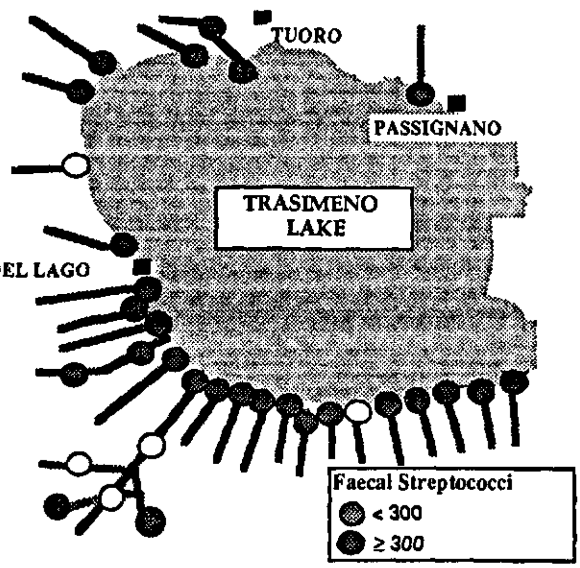


Fig. 4 - Faecal contamination of lake tributaries registered in 1996.

It is evident that the faecal loads entering in the lake are important, being the positive samples 21 on 32 for faecal coliforms and 27 on 32 for faecal streptococci, respectively. Adding the faecal loads discharged from the wastewater treatment plants (Fig. 5), it is obvious that heavy organic loads are continuosly let in the Trasimeno.

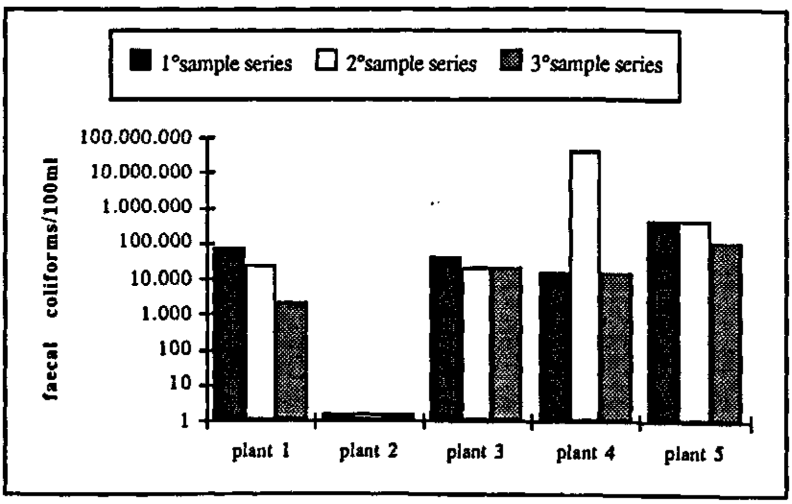


Fig. 5 - Faecal coliform amounts coming from the treatment plants observed in 1996.

Macroinvertebrate community observations in some of the biggest creeks confirmed the bad quality of the waters put in the lake (Fig. 6).

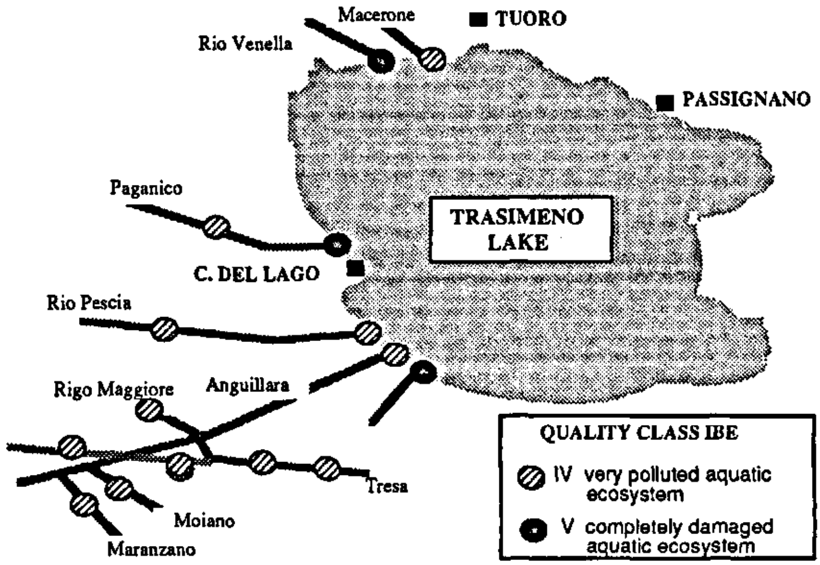


Fig. 6 - Macroinvertebrate biotic index (Ghetti 1995) values pointed out in some ditches in 1996.

Furthermore, the efficiency of treatment plants discharging in the lake resulted very low (Cingolani & Ciccarelli, 1993; Cingolani, in press), beeing the plants regularly in bulking or in pin-point. Such phenomena were ignored by local administrators because of the low concentrations of faecal microorganisms, nitrates and phosphates in the lake water (in consequence of high pH, organic matter sedimentation, macrophyte and alga capture), the only indicators considered. But other signals appeared, pointing out an increasing eutrophication: the progressive decrease of transparance, pH and D.O. high values, the increase of blue algal blooms (*Aphanizomenon flos-aquae*, *Anabaenopsis circularis*, *Anabaena* spp, *Phormidium* spp) and chlorophyll-a (Fig. 7 and 8), always more fastidious mosquito invasions (Anopheles spp, *Chironomus* spp, *Simulium* spp,).

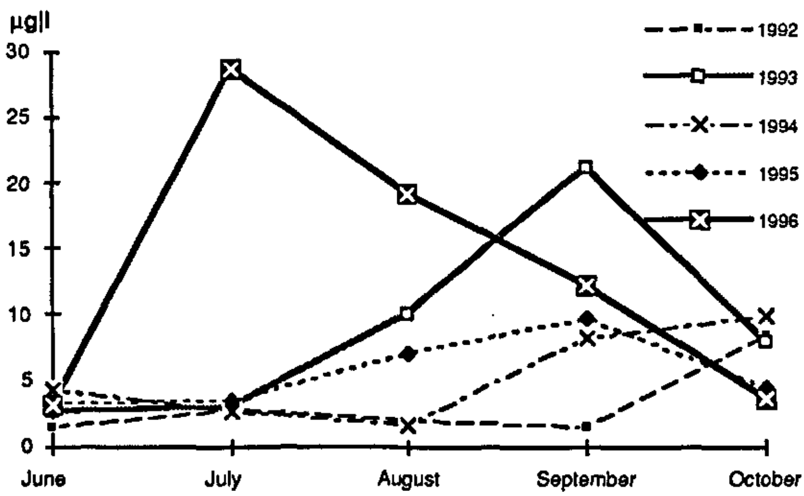


Fig. 7 - Monthly averages of chlorophyll values detected from 1992 to 1996.

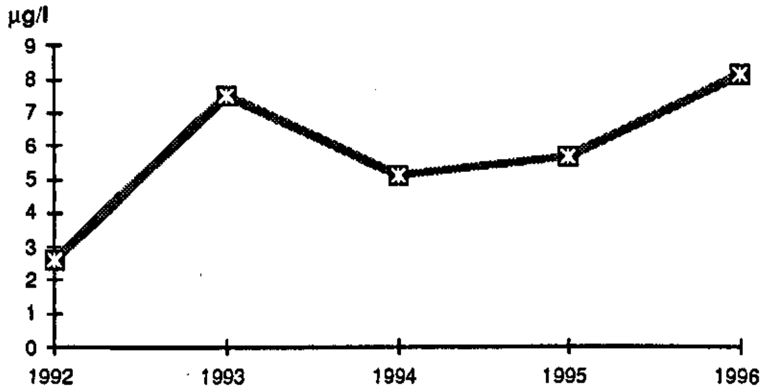


Fig.8 - Trend of the clorophyll concentrations in the years (annual averages).

Furthermore, the lake was colonized from immemorial time by wide meadows of submerged macrophytes for all the thickness of water.
From some years the popolations of *Nimphaea alba, Nuphar luteum and Ranunculus aquatilis* are completely disappeared, while now the composition of aquatic biocenosis is limited to the presence of a large amount of *Potamogeton perfoliatus*, followed by *Miriophyllum spicatum* and *Ceratophyllum demersum* (Granetti, 1966).
In the last years Districtual Administration has increased the cut of macrophytes, in consequence of the purchase of aquatic grass cutting boats and of the purpose to use macrophyte in agricolture after a compost process.
It is our opinion that this behaviour might have favoured the increase of phytoplancton to the detriment of the macrophyte grow. In summer 1996 phytoplancton colonized all the water thickness, while the chlorophyll values reached the maximum of the concentrations registered until now (peaks of 44,9 µgll).
The blue-algae bloom, from the end of June to the end of September, showed species capable to produce phytotoxins (Harada et al. 1991; Sivonen et al. 1992).
In this period the stink of drinking water taken from the lake was terrible and many people brougth actions against the mayor.
The municipal administration decided, finally, to add active carbon filters to the existing sand filter (only for some months).
Already in the last year our laboratory pointed out that the sand filter was obsolete and inadequate, having found a sludge stratum full of diatom valves on the walls of a reservoir.

The worrying grow of blue algae capable to release tumor promoter toxins should induce the public administrations and the local sanitary unit to promove immediately safeguard actions, in order to avoid risk for people and aquatic life.
The logic think to do, for having a lasting restoration of water and environment, should be the deviation of the organic loads from the lake.
Such operations should have beneficial effects on the control of chironomide and simulide populations, microalga eaters (Arshad, 1991), increasing the turist flow in consequence.The purchase of yellow panels adopted by the municipal administration was, in fact, unsuccessful for attracting mosquitos.

CONCLUSIONS

In conclusion, we will underline how simple observations on what is happening and just a little good sense might avoid that the lake eutrophication goes too far, causing health and pecuniary heavy damages for population. The lake people must to know that a low quality water may damage also the same activities which have polluted. The linking of other streems shall be surely insufficient to contein the undesirable effect of eutrofication, even if the natural water turnover time (25 years) may be reached. The construction of purification plants to treat factory wastewaters, as suggested by some researcers, is absolutely inadequate if the effluents are employed in fertirrigation (too much factories) or, worse still, discharged directly into the lake. The administrations had to have cleared the lake fate many and many years ago, and then to take the consequent decisions and actions. Surely, the insertion of so many factories in a so narrow watershed, unitely to the discharge of civil wastewater (even if submitted to purification treatment) in the lake have been the worse decision taken from the local administration.

REFERENCES

Arshad A. (1991). Perspectives on Management of Pestiferous Chironomidae (Diptera), an Emerging Global Problem. *Journal of the American Mosquito Control Association* 7, 2: 260-281.

Bourelly P. (1985). *Les algues d' eaou douce.* Edition N. Boubée, Paris.

Cingolani L., Ciccarelli E.(1993). Relationships between protozoan community dynamics and performance of the activated sludge process in wastewater treatment plants.*The Journal of Eukaryotic Microbiology*, 4: 26.

Cingolani L. (in press). Considerations On The Role Of Some Ciliate Protozoans In The Activated Sludge. *2nd International Conference on "Microorganisms in Activated Sludge and Biofilm Processes*, International Association Water Quality, Berkeley 21-23 July 1997.

Ghetti P.F. (1995). Indice Biotico Esteso. - *Notiziario dei metodi analitici. ISSN:0392-1425*.IRSA CNR.

Decreto Ministeriale 16.6.88

Ghetti P.F. (1995). Indice Biotico Esteso. - *Notiziario dei metodi analitici. ISSN:0392-1425*.IRSA CNR.

Granetti B. (1966). La flora e la vegetazione del lago Trasimeno. *Idrobiologia*. 4: 115-182

Harada K.I., Ogawa K, KimuraY., Murata H., Suzuki M, Thorn P.M., Evans W.R., Charmichael W.W. (1991). Microcystins from *Anabaena flos-aquae* NRC 525-17 *Chem. Res. Toxicol.* 4: 535-540.

Istituto Ricerca Sulle Acque (1995). *Metodi Analitici Per Le Acque.* Istituto Poligrafico e Zecca dello Stato Ed., Roma, pp 318-327.

Regione Umbria (1995). *Piano Urbanistico Territoriale.*

Sivonen K., Skulberg O.M., Namikoshi M., Evans W.R., Charmicael W.W. Rinehart K.L. (1992). Two methil ester Derivatives of Microcystins, Cyclic Heptapeptides Hepatotoxins, Isolated from *Anabaena flos-aquae* CYA 83/1. *Toxicon*. 30: 1465-1471.

Bioavailability of Phosphorus as a Tool for efficient P Reduction Schemes

Peter Gerdes and Sabine Kunst

Institute for Sanitary Engineering and Solid Waste Management, University of Hannover, Welfengarten 1, 30167 Hannover, Germany

ABSTRACT

The bioavailability of Phosphorus from different sources has been evaluated in the catchment area of the River *Ilmenau* (Lower-Saxony, Germany) by using algal assays. The P bioavailability describes the different potential of P from various sources of supporting eutrophication. Effluents from sewage treatment plants were highly bioavailable (72 % of TP) whereas rainwater (26 %) and erosion effluents (30 %) showed a low bioavailability. In order to develop effective strategies to minimize P inputs into the river, source specific P bioavailability indices were determined and combined with a P balance to calculate inputs of bioavailable P (BAP) instead of total P (TP). It could be shown that the relative importance of the different P sources changes when applying BAP. Measures to reduce P inputs into the River *Ilmenau* will take P bioavailability into consideration and therefore lead to a more cost-effective management.

KEYWORDS

Algae; bioavailability; eutrophication; phosphorus; reduction scheme.

INTRODUCTION

Phosphorus is considered to be one of the major determinants of eutrophication and has been the focus of worldwide efforts to improve surface water quality. Phosphorus compounds enter the rivers in different forms and compositions, depending mainly on their origin. Algal growth is supported when the effluents reach the big rivers and the sea. Although a direct uptake of some organic phosphates cannot be excluded the main P source for planktonic algae is orthophosphate. Bioavailable P (BAP) is defined as the sum of immediately available P and P that can be transformed into an available form by naturally occurring processes (Boström et al., 1988). Depending on the source specific composition of the effluents, the directly available and potentialy available portion of total P and therefore their potential ecological impact differ widely.

In the present economic situation P management strategies should be geared to provide maximum benefit at least cost. In order to achieve an effective control of eutrophication the input of especially high and fast available P forms should be reduced. These forms support algal growth more than low and slowly available forms.

So far calculations within reduction schemes to control P inputs are mainly based on total P (Clesceri et al., 1986; Cluis et al., 1988; Behrendt, 1993). The potential ecological impact of the various P forms has not been taken into consideration yet. To increase the efficiency of reduction schemes it would be useful to classify the P sources with respect to their P bioavailability. The determination of a source specific P bioavailability (Källqvist and Berge 1990; Sharpley and Smith, 1993) forms the basis for this.

Different chemical extractions have been used to define bioavailable P but of all methods the algal assay (AGP-Test) gives the most reliable information about the portion of total P bioavailable (Dorich et al., 1980; Young & Depinto, 1982; Sonzogni et al., 1982, Boström et al., 1988; Sharpley and Smith, 1994). The upper limit of the amount of potential bioavailable P can be evaluated with this method.

The aim of this study was to develop a technique through which the potential impact of different P forms can be taken into account when planing measures to reduce P inputs into surface waters. Therefore the calculated annual export of total P from a defined catchment area was combined with P bioavailability indices for the major sources, derived from algal assays.

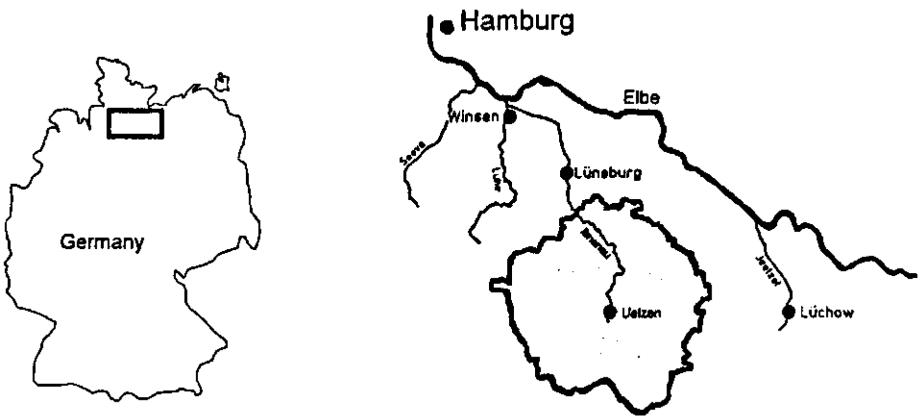


Figure1. The study catchment area of *Ilmenau*

MATERIALS AND METHODS

Sampling and sample pretreatment.

Samples of the important P sources were collected in the catchment area of the River *Ilmenau* in Northern Germany (Fig. 1) in 1995 and 1996. This area consists predominantly of sandy soils and shows the following land use data: arable land 43 %, forestry 33 %, grassland 18 %, settlements 5 %. The size of the study area is about 140.479 ha. Water samples were transported to the laboratory in a cooling box immediately and stored in the freezer at -20°C for at least one day and not longer than one week. Indigenous algae could be removed with this treatment. The sterilisation of the samples by autoclaving, filtration or gamma-radiation caused a massive change in the chemical composition of the samples especially regarding the different P fractions. The problem of sample pretreatment was often discussed when AGP-tests were conducted (Filip and Middlebrooks, 1973; Klapwijk et al., 1989; Källqvist and Berge 1990) and similar results were reported. For this reason an approach without complete sterilisation was chosen in this study. Deep frozen samples showed no significant change in the different P fractions and a mean reduction of 50 % of the bacterial contamination was achieved. A significant nutrient competition between bacteria and algae was observed only in samples from sewage treatment plants with a high content of BOD_5. These samples were not used for further evaluation.

Surface soil samples were collected from potentially erosion sensitive fields all over the catchment area. These samples were air dried and sieved at 315 µm in order to mimic the separation of smaller particles during erosion runoff. The soil samples were diluted in 10 % P-free nutrient medium to a concentration range between 100 µg P/l and 200 µg P/l. A water/soil ratio between 2000:1 and 5000:1 could be obtained with this method.

The bioavailability of organic and inorganic fertilizers was evaluated by diluting liquid manure and solid fertilizer in 10 % P-free nutrient growth medium.

Assessment of bioavailable phosphorus (BAP)

All samples were analysed for total phosphorus (TP), inorganic phosphorus (IP) and organic phosphorus (OP) according to Krogstad and Lovstad (1991). The samples were diluted with distilled water when the concentration of TP was above 200 µg/l. All samples were diluted with a P-free Sce medium (Bringmann and Kühn, 1978) to a final concentration of 10 % Sce medium. This was to ensure that P is the growth limiting nutrient. 20 ml of the samples were inoculated with $3,4 \times 10^6$ cells of *Scenedesmus subspicatus* in 50 ml Erlenmeyer flasks. The flasks were closed with cotton stoppers and incubated on a shaker at 23°C under continuous illumination from cool-white fluorescent tubes. The algal biomass was analysed after the growth ceased, usually after 9-14 days. The biomass was analysed photometrically with a spectral photometer. The measured extinction was translated to cell number via correlations obtained with manual cell counts. The BAP was measured from the yield in the cultures using standard curves obtained under identical conditions with different concentrations of orthophosphate in 10 % Sce-medium, assuming that this P-source is 100 % available. A linear relationship between algal growth and P concentration was observed in a range between 10 µgP/l and 200 µgP/l. In accordance with Klapwijk et al. (1982) the standard cultures were always incubated simultaneously with the assay cultures. A growth inhibition by toxic compounds was detected following the method of Källqvist and Berge (1990). Samples with toxic effects were not used for further evaluation. All tests were carried out in triplicate except the soil samples with 5 parallels. BAP was calculated according to the standard curves and the terminal yield in the samples. The results were documented as µg BAP/l or % BAP of TP:

$$\text{BAP [µgP/l]} = \text{Biomass } [10^9 \text{ cells/l}]/0{,}0431$$

$$\text{BAP [\%]} = (\text{BAP[µgP/l]} \times 100)/\text{TP [µgP/l]}$$

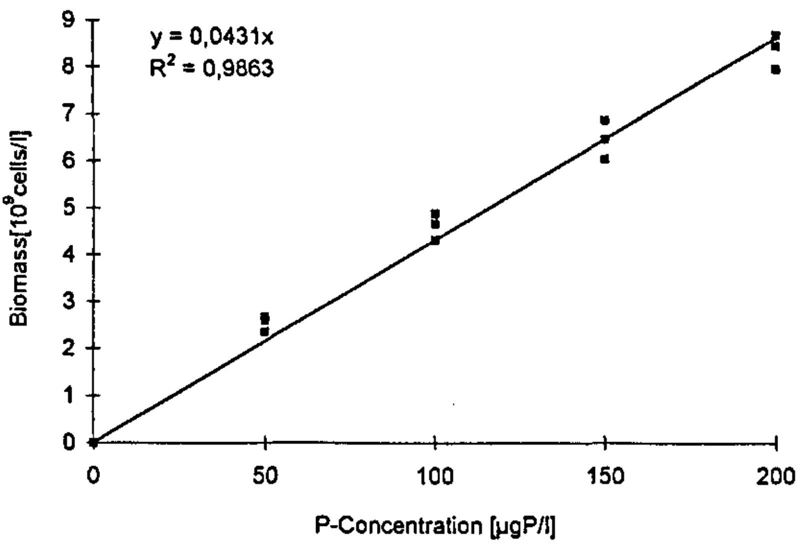


Figure 2. Cell yield of *Scenedesmus subspicatus* as a function of orthophosphate-P concentration in growth medium 10 % Sce

Application of P bioavailability indices

In the framework of the project "*Economic efficiency control of water protection actions in the European Community*" a phosphorus balance was carried out for the catchment area of the River *Ilmenau*. After identifying all P sources, the annual TP input from each source was calculated. All TP inputs were added to calculate the total annual load leaving this catchment area. The aim of these calculations is to find the most effective P reduction strategies in respect of cost-benefit (Fehr, 1996). These results were verified with measurements in the river (mean value for the years 1989-94). To finally calculate the inputs of bioavailable P (BAP) instead of total P (TP), the source specific P bioavailability was measured for the important sources with the methods described above (BAP as % of TP, Bioavailability of for example 50 % gives a source specific bioavailability index of 0,5). The specific values were obtained by using the mean value of all individual samples analysed for one source. To verify the results, the measurements of BAP were also conducted with river samples (monthly sampling within 1996). The results of BAP for each source were then linked to the TP calculations to calculate BAP instead of TP. Therefore the TP values for each calculation category or sub-category were multiplied with the corresponding bioavailability indices.

RESULTS AND DISCUSSION

Source specific P bioavailability

The results of the BAP analysis of all sample categories (Table 1) indicate that large differences regarding the algal availability of effluents from different sources exist. The lowest bioavailability could be observed in rainwater samples (26 %), probably due to the high content of the particulate P fraction (dust). Little information exist on the bioavailability of P from this source. According to Sonzogni et al. (1982) about 25-50 % of TP is available, whereas the differences are based largely on the different content of dissolved P.

Sample category	n	TP [µg/l]	BAP [%]			Bioavailability Index
			min	mean	max	
Sewage treatment plants	39	1422,0	45	72	98	0,72
Anorganic fertilizer	3	45350*	39	63	87	0,63
Groundwater	3	339,5	41	54	77	0,54
Grassland drainage water	29	133,1	19	54	94	0,54
Municipal surface runoff	46	308,5	20	53	94	0,53
Manure	4	41000	25	52	68	0,52
Arable land drainage water	39	122,4	4,5	43	97	0,43
Agricultural surface soil	24	705,0*	9	30	57	0,30
Rainwater	4	66,9	19	26	33	0,26

* TP [µg/g dry matter]

Table 1. Mean values for TP, BAP and bioavailability indices for different categories of samples

Erosion effluents as simulated by diluting surface soil samples (not recently fertilised) with nutrient medium also have a low BAP value (30 %). The bioavailability of the particulate P derived from this sources is similar to the bioavailability of tributary particulate P as defined by several investigations (Sonzogni et al, 1980). Dorich et al. (1980) reported an average of 20 % of the particulate P from agricultural runoff to be bioavailable. Similar values were reported by Young and Depinto (1982), who observed a bioavailability between 6,1 and 35,8 % of tributary particulate phosphorus, and Källqvist and Berge (1990), who reported an average of 20 % of natural erosion P to be bioavailable. In accordance with Young and Depinto (1982) and Sonzogni et al. (1982), it was noticed in all analyses that a high content of particulate P causes a low bioavailability.

The results obtained with non fertilized soils were compared with analysis from recently fertilized soils (anorganic fertilizer). A bioavailability of 59 % could be observed, which is twice the value of non fertilized soils. These findings indicate that erosion runoff in times of fertilization has a much higher potential of supporting algal growth then in non fertilized times.

The results of analysis of the different sewage treatment plants were sorted according to the type of P elimination used (Fig.3). It could be shown that the biological or chemical P elimination minimizes especially the high bioavailable P fractions in the effluents. The highest bioavailable P value was found in effluents from treatment plants without any P elimination (87 %). Treatment plants with biological or chemical P elimination showed a portion of 67 % P available, respectively. In general, a P elimination seems to be very useful in removing especially high available P compounds.

An average value of 72 % P bioavailability was calculated for the sewage treatment plants of the study catchment area. A specific investigation on the bioavailability of P in municipal effluents (Young and Depinto,1980) showed similar results. Also based on algal bioassays, bioavailable P was estimated to be about 70 % of TP in treated effluents.

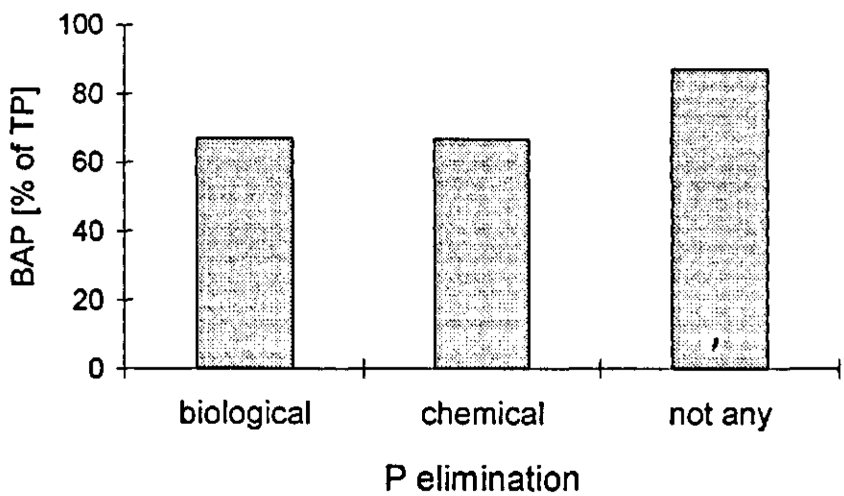


Figure 3. P bioavailability of sewage treatment plant effluents according to the type of P elimination

The average bioavailability of agricultural drainage effluents was between 43 % for grassland and 54 % for arable land respectively. Urban drainage effluents had a BAP of 53 %. Liquid manure had a BAP of 52 % which was probably due to the high content of organic P (56 %). This P fraction is considered to be less bioavailable in comparison with anorganic P compounds (Hegemann et al., 1982; Boström et al, 1988). The analysed solid fertilizers had a mean BAP of 63 %, mainly dependent on their solubility.

Calculation of bioavailable P inputs

The obtained bioavailability indices (Table 1) were linked to the P balance for the catchment area of *Ilmenau* to finally calculate BAP instead of TP (Table 2). Wherever possible, the indices were directly applied for the analysed source (e.g. sewage plants, municipal surface runoff). The index for interflow (no analysis) was calculated as average value between agricultural drainage water and groundwater. Direct inputs were calculated as the sum of rainwater, organic and anorganic fertilizers and inputs from farms. The agricultural drainages consists of arable land drainages and grassland drainages. Bioavailability indices were applied for all sub-categories.

Balance calculation categories	inputs [t/a] TP	BAP
Total	93,43	49,52
Sewage treatment plants	22,65	16,31
Urban runoff	14,75	7,82
Direct inputs	31,92	16,18
Erosion effluents	15,24	4,57
Interflow	2,21	1,13
Agricultural drainage	1,14	0,53
Groundwater	5,51	2,97

Table 2. Annual TP and BAP inputs from the catchment area of *Ilmenau* (1989-94)

The comparison between the annual TP inputs and the calculated BAP inputs (Figure 4) indicates a shift in the relative importance of the single sources when using BAP as parameter within P reduction schemes. The relative importance of the sewage treatment plant effluents increases, whereas the role of erosion effluents decreases. The importance of using BAP is even bigger when catchment areas with a higher portion of inputs from sewage treatment plants are examined. Although the P inputs from this source were reduced dramatically in the past decade, the significance of municipal effluents for supporting eutrophication is still obvious.

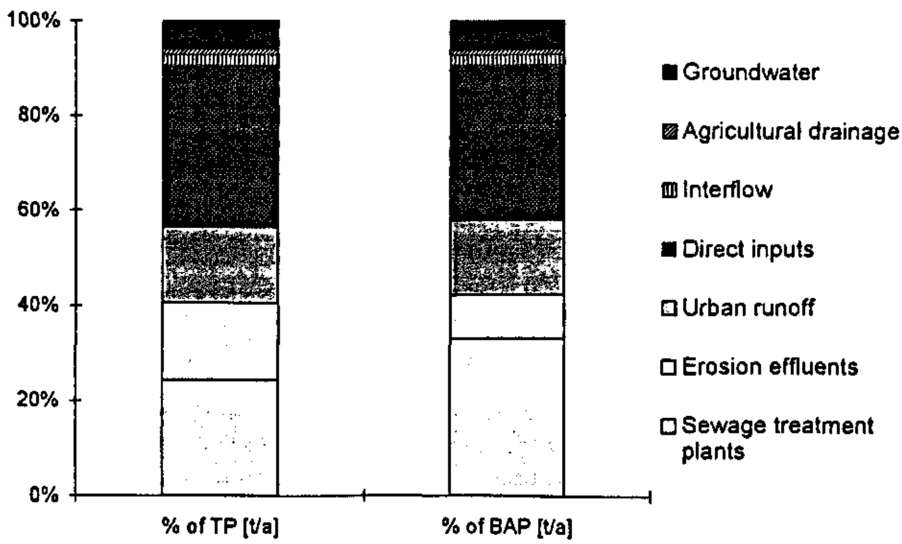


Figure 4. Relative importance of different P sources. Comparison of annual TP and BAP inputs into the River *Ilmenau* (1989-94), expressed as % of the total TP and total BAP input

The mean bioavailability as average for all inputs was 53 %. This value could be confirmed with measurements in the River *Ilmenau* at the outlet of the catchment area. The average BAP of monthly taken samples in 1996 was 50 %. Significant seasonal differences in the BAP value could not be observed.

The development of different management strategies to reduce P inputs into the River *Ilmenau* will be conducted on the basis of BAP. The choice of different reduction measures will be influenced by the potential ecological impact of the reduced source upon eutrophication. Hence the use of BAP will increase the efficiency of the reduction of P inputs.

CONCLUSIONS

This study has shown that the P bioavailability differs widely depending on the source of the P input. This source specific bioavailability cannot properly be evaluated by chemical analysis. With an algal assay it is possible to measure the maximum algal availability . As a result the potential effect of different P species from different sources upon eutrophication can be evaluated. With the use of this tool within P reduction schemes it is possible to develop strategies to reduce the input of particulary those P species with a high impact upon eutrophication. A reduction of bioavailable P inputs in the *Ilmenau* catchment area will lead to a reduction of algal biomass in the River *Elbe* and in the *North-Sea.* The combination of biological survey with balance calculations integrates biological processes in applicable management strategies. Within watershed management programmes as proposed by the new framework directive on water of the European Community (Commission of the European Community 1997), the use of BAP could help to develop efficient strategies to minimize eutrophication in Europe.

REFERENCES

Behrendt, H. (1993). Separation of point and diffuse loads of pollutants using monitoring data of rivers. *Wat.Sci.Tech*, **28**, 165-175

Boström, B., Persson, G. and Broberg, B. (1988). Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiologia*, **170**, 155-155

Bringmann and Kühn, (1978). Veränderungen der biologisch wirksamen Beschaffenheit der Sedimente im Fließgebiet des Teltowkanals durch Einleitung kommunaler Abwässer. *Zeitschrift für Wasser- und Abwasserforschung*, **11** (3/4), 87-94

Cluis, D., Couture, P., Bégin, R. and Visser, S.A. (1988). Potential eutrophication assessment in rivers; relationship between produced and exported loads. *Schweiz. Z. Hydrol.*, **50** (2), 166-181

Clesceri, N.L., Curran, S.J. and Sedlak, R.I. (1986). Nutrient loads to Wisconsin lakes: Part 1. Nitrogen and Phosphorus export coefficients. *Water Res. Bull.*, **22** (6), 983-990

Commission of the European Community, (1997). Proposal for a council directive establishing a framework for community actions in the field of water policy. COM(97) 49 final, 97/0067(SYN)

Dorich, R.A., Nelson, D.W. and Sommers, L.E. (1980). Algal availability of sediment phosphorus in drainage waters of the Black Creek Watershed. *J. Environ. Qual.*, **9** (4), 557-562

Fehr, G. (1996). Effektivitätskontrolle ökonomischer Steuerungsinstrumente im Gewässerschutz. *Wasser und Boden*, **48** (1), 45-50

Hegeman, D.A., Johnson, A.H. and Keenan, J.D. (1983). Determination of algal-available phosphorus on soil and sediment: a review and analysis. *J. Environ. Qual.*, **12** (1), 12-16

Källqvist, T. and Berge, D. (1990). Biological availability of phosphorus in agricultural runoff compared to other phosphorus sources. *Verh. Internat. Verein. Limnol.*, **24**, 214-217.

Klapwijk, S.P. and Kroon, J.M.W. and Meijer, M-L. (1982). Available phosphorus in lake sediments in the Netherlands. *Hydrobiologia*, **92**, 491-500

Klapwijk, S.P. and Bolier, G. and van der Does, J., (1989). The application of algal growth potential tests (AGP) to the canals and lakes of western Netherlands. *Hydrobiologia*, **188/189**, 189-199

Krogstad, T. and Løvstad, Ø. (1991). Available soil phosphorus for planktonic blue-green algae in eutrophic lake water samples. *Arch. Hydrobiol.*, **122**, 117-128

Filip, D.S. and Middlebrooks E.J. (1975). Evaluation of sample preparation techniques for algal bioassays. *Wat.Res.* 9, 581-585

Sharpley, A.N. and Smith, S.J. (1993). Application of phosphorus bioavailability indices to agricultural runoff and soils. In: *Application of Agricultural Analysis in Environmental Studies*, K. B. Hoddinott and T.A. O`Shay (Ed.), American Society for Testing and materials, Philadelphia, pp 43-57

Sonzogni, W.C., Jeffs, D.N., Konrad, J.C., Robinson, J.B., Chester, G., Coote, D.R. and Ostry R.C. (1980). Pollution from land runoff. *Environ. Sci. Technol.*, **14**, 148-153

Sonzogni, W.C., Chapra, S.C., Amstrong, D.E. and Logan, T.J. (1982). Bioavailability of phosphorus inputs to lakes. *J. Environ. Qual.*, **11** (4), 555-563

Young, T.C. and De Pinto, J.V. (1982). Algal-availability of particulate phosphorus from diffuse and point sources in the lower Great Lakes basin. *Hydrobiologia*, **91**, 111-119