Hydrophysical and Ecological Models of Shallow Lakes and Reservoirs

Summary Report of an IIASA Workshop
April 11–14, 1978

Sven E. Jørgenson, Editor

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Sven E. Jørgensen, Editor
The Royal Danish School of Pharmacy
Copenhagen, Denmark

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INTERNATIONAL INSTITUTE FOR APPLIED SYSTEMS ANALYSIS
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Views expressed herein are those of the contributors and not necessarily those of the International Institute for Applied Systems Analysis.

PREFACE

An important subtask within the Resources and Environment Area (REN) of the International Institute for Applied Systems Analysis (IIASA) is the development and application of models for environmental quality control and management. The initial objectives of this task are to assess existing models, to develop improved hydrophysical and ecological models as tools for the analysis of water quality problems, and to apply these models to lakes, reservoirs, and river systems.

On September 13-16, 1977, a workshop on the general aspects of water quality modeling (IIASA CP-78-10) was held at IIASA. The most recent theoretical developments in the field of Water Quality were discussed at this meeting. At the same time the application of hydrophysical and ecological models to various water bodies was considered. As a result of this workshop it was decided that attention should be focussed on the water quality problems of natural lakes and man-made impoundments (reservoirs). In addition, it was felt that IIASA could make an important contribution to the use of models for water quality control and management purposes by attempting to bridge the gap between the hydrophysical and ecological modeling disciplines. Accordingly, it was decided to convene two specialized workshops. The first of these was on Hydrophysical and Ecological Modelling of Deep Lakes and Reservoirs (IIASA CP-78-7) held on December 12-15, 1977, in Laxenburg. The second of these was on hydrological and ecological modeling of shallow lakes and reservoirs, and was held on April 11-14, 1978, in Laxenburg.

The subject of deep lakes and reservoirs chosen for the December workshop implies a basic concern with stratification and interactions at the free surface boundary rather than with conditions of full vertical mixing and bottom sediment interactions.

The subject of shallow lakes and reservoirs chosen for the April workshop implies a basic concern with the condition of full vertical mixing and bottom sediment interactions. This report summarizes the results of this second workshop, which was attended by 28 people representing 14 countries and by 13 IIASA staff members from 5 countries. Prior to the workshop, a set of questions related to ecological and hydrophysical modeling problems were formulated by the IIASA staff and sent to the participants with the request to indicate on which questions they would like to give a short presentation as a start to the discussion of the questions. Topics discussed at the workshop included:

-- *Characteristic Features of Shallow Lakes and Reservoirs:* influence of wind and wave action, longitudinal and vertical mass transport processes, exchange of nutrients between the water body and the sediments, influence of sediment types on the transformation processes of chemical compounds
-- *Hydrophysical Models:* horizontal and vertical transport and diffusion processes, interaction across the water/sediment interface
-- *Ecological Models:* evaluation of available data by simple models, sensitivity analysis, improvement in the quantity of data, further development of ecological models of shallow lakes taking into account the binding and mobilization of nutrients in the sediments
-- *Water Quality Models:* limiting nutrients/carbon, nitrogen, phosphorus, element-cycle models/simple and complex models
-- *Field Data Collection and Model Verification Techniques:* coordination of chemical and biological field measurements with water quality models, choice of parameters
-- *Possible Case Studies*

These discussions are summarized in the Introduction.

In addition, some of the participants presented original papers at the discussion of the ecological and hydrophysical questions. These papers are included in the second part of this Proceedings following the Discussion. It should be noted that a paper by H.A. Tsvetova on Numerical Modeling of the Dynamics of Lake Baikal was presented at the workshop. It is not included in these proceedings but will be published separately in IIASA's Professional Paper Series. The paper was originally intended for presentation at the workshop on Hydrophysical and Ecological Modelling of Deep Lakes and Reservoirs.

This workshop was an excellent opportunity for experienced modelers from both the ecological and hydrophysical areas to come together for a discussion of problems of mutual interest and to exchange ideas and viewpoints.

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CONTENTS

INTRODUCTION

S.E. JØrgensen

There are many specialized models for solving eutrophication problems. They range from those in which one to four nutrients may be considered, to those in which one or several layers and segments may be considered. Some models are based on constant stoichiometry, others on independent nutrient cycles. Unfortunately most of the models have not been validated or calibrated.

No universal model exists for solving the above problems. A particular model is selected on an individual basis according to the nature of the problem to be solved, and most models are based on biological/chemical knowledge of the processes taking place in the lake. A black box approach seems to be inappropriate for modeling a lake system, because it does not provide general information about the aquatic system; but experience indicates that a multidisciplinary scientific approach is required.

Many models are limited by the lack of suitable field data for model calibration and verification. Therefore, in the model, the description of an ecosystem is based upon the data, and thus an accurate description of the ecosystem is dependent upon the accuracy of the data (e.g., measurement error levels, sampling accuracy). These general items will be mentioned briefly in several of the questions listed in the agenda of this workshop.

What is the state of the art of lake modeling today? Using a universal model, it is impossible to find an improved scientific description of a specific process in an aquatic ecosystem, but it is possible to obtain an improved description of a specific process by using a submodel based on intensive *in situ* measurements, or by examining an individual biological/chemical process in the laboratory. This does not mean that universal models are not useful--they can be used to examine the overall behavior of several interacting processes, and may therefore be necessary for management purposes.

Submodels, however, are more useful as a scientific tool. The selection of an appropriate model is dependent upon finding the correct balance of information, for example, all important processes must be included, but not every detail of less important processes. If too many state variables are included the model will be unwieldy, and if too many parameter values are introduced, this could lead to inaccuracies. But, in contrast, a model that is too simple might not describe adequately the dynamics of the system.

No general principles underlying the reactions of ecosystems have been formulated, and a more reductionist approach to modeling does not seem to be feasible. Instead a holistic approach embodying a set of ecological principles is needed. The description of current ecological systems can be compared to that of gases in a room, by means of velocity, direction, and groups of molecules. In the future, ecological systems will be expressed by general laws analogous to those of gases, and will contain terms equivalent to Avogadro's number and the like. These and other fundamental problems of current water quality modeling will be discussed in the course of these Workshop Proceedings.

Part I

DISCUSSION

DISCUSSION OF QUESTIONS RELATED TO HYDROPHYSICAL AND ECOLOGICAL MODELING OF SHALLOW LAKES AND RESERVOIRS

A. ECOLOGICAL TOPICS

1. *The validity of constant stoichiometric models versus element-cycle models. How many element cycles are necessary for a eutrophication model?*

Reported by G. van Straten

In general a preference was shown for element-cycle models or at least for variable stoichiometric models, in order to predict algal bloom. This is because each of the nutrients--phosphorus, nitrogen, and silica--is to some extent cycled independently, owing to such phenomena as luxury uptake. The use of an element-cycle type of model would seem to be essential in the following two instances:

-- If a switch occurs during the season, whereby the limiting factor passes from one nutrient to another (e.g., luxury phosphorus uptake, when silica is the limiting nutrient)
-- If the internal nutrient cycle is significant in comparison with the external loading (internal cycling in most cases is more important for shallow lakes than for deep lakes)

In addition, the question was raised whether the phosphorus/nitrogen ratio in the cell had some influence on the settling rate. If this is the case, then a constant stoichiometric model cannot be used. In some cases only an independent element-cycle model is able to predict accurately the time or the peak height of an algal bloom maximum. However, some simulation results showing only a minor difference between the two types of models have been reported. This minor difference is more apparent if the changes in phytoplankton concentrations are small. A serious disadvantage of the independent element-cycle model is that it requires the introduction of more parameters into the model. Therefore in certain cases it may be beneficial to use a constant stoichiometric model together with measured values for the biomass/nutrient ratio. The best example of this is in the design of an algal growth basin to be operated under steady conditions.

However, many of the parameters in the independent element-cycle models are well known, such as the minimum and maximum concentration of phosphorus, nitrogen, carbon, and silica. Calibration can thus be carried out for restricted parameter intervals, counterbalancing the disadvantage of having more parameters in the model.

The conclusion of this discussion is that constant stoichiometric models may be used to obtain a reasonable approximation of the time or peak height of algal bloom, providing that no great changes in intercellular biomass nutrient ratio are to be expected. But a detailed and more accurate description can only be obtained from a nutrient-cycle type of model at the expense of introducing more parameters, most of which are approximately known, but require accurate estimation.

2. *Which equations are most suitable for describing the nutrient uptake rates by phytoplankton of (a) phosphorus, (b) nitrogen (NH_4^+, NO_2^- as well as NO_3^-), (c) carbon, (d) silica? How should the growth and mortality of phytoplankton with single and multiple cellular concentration of nutrients be modeled? Which light and temperature expressions are suitable?*

Reported by M.W. Lorenzen

Equations (1), (2), and (3) for nutrient uptake in phytoplankton growth were presented:

$$\text{growth} = \mu max \cdot f(T) \cdot f(PA) \cdot f(NA) \cdot f(CA) \cdot f(A) \quad , \tag{1}$$

where A refers to algal concentration, and T, P, N, and C are temperature, phosphorus, nitrogen, and carbon. The nutrient uptake--e.g., for phosphorus--can be described by Equation (2).

$$UP = UP_{max} \frac{PA_{max} \cdot PHYT - PA}{PA_{max} \cdot PHYT - PA_{min} \cdot PHYT} \cdot PHYT \cdot \frac{PS}{KP + PS} \quad , \tag{2}$$

where S represents dissolved nutrient.

A differentiation between ammonia and nitrate uptake was included by the use of Equation (3):

$$UN = UN(Nit) + UN(NH_4^+) \quad ,$$

$$UN(Nit) = UN_{min}(Nit) + \left(1 - \frac{Nit}{NH_4^+}\right)(UN_{max}(Nit) - UN_{min}(Nit)) \quad . \tag{3}$$

The effect of temperature was presented as the functional relationship shown in Figure 1. For this type of function it is recommended that a look-up table is employed in order to minimize computer time.

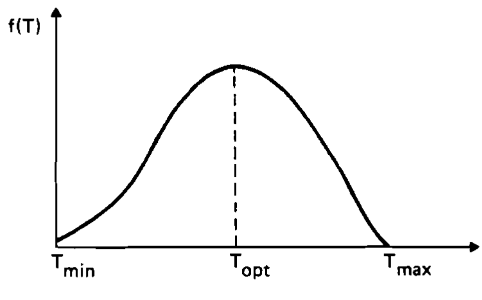


Figure 1. Phytoplankton growth, f(T), as a function of temperature, T.

It is recommended that these expressions should be used in situations discussed in conjunction with question A1. The Michaelis-Menten expression, which is often used, shows, however, little difference from the more detailed description referred to above when short-term batch culture experiments are interpreted.

It has been suggested that regression analysis can be used to lump together unknown parameter values when long-term data are available. This method can only be recommended for simple cases. It was also stated that when combining a number of limiting terms (nitrogen, phosphorus, temperature, light, silica) a minimum or threshold approach is superior to a multiplicative equation, see above.

However, the whole process of nutrient uptake and growth is more complex and difficult; it seems necessary to assume that parameters change with the temperature. The same is true for the "Michaelis-Menten constants", since they change with light intensity. Unfortunately, there are at present insufficient data to construct a response function for phytoplankton growth as a function of temperature, nitrogen, phosphorus, silica, etc. A schematic presentation of the values of the various rate-limiting terms throughout an annual cycle might clarify their relative importance at different times of the year. The question "How do we take the adaptation of phytoplankton into account?" merits further discussion.

3. *Which equation is relevant for describing the grazing rate? Should more than one species of zooplankton be included?*

Reported by D.O. Logofet

There are several types of predator-function responses to an increasing prey density; these include the prey/predator system of phytoplankton/zooplankton of particular interest here.

It is possible to select a model type by means of statistical procedures of hypothesis testing. In the simplest case of distinguishing between two analytical expressions, f_1 and f_2, the dichotomy problem takes the following form:

$$f = \Theta f_1 + (1 - \Theta) f_2 \quad , \tag{4}$$

the hypotheses H_1 and H_2 being

$$H_1 : \Theta = 1, \; H_2 : \Theta = 0 \quad . \tag{5}$$

However, in complex simulation models with more than one state variable for both the phytoplankton and zooplankton species, the results often appear to be insensitive to variations of the grazing-rate parameters. From a theoretical point of view this insensitivity is caused by the data being insufficient to allow for a distinction to be made between two or more hypotheses. From an ecological point of view it is worth while to investigate the means to incorporate into the model a few varieties of phytoplankton and at least two varieties of zooplankton. But for management purposes often only an average situation is of interest, and in this case the modified Michaelis-Menten expression might suffice. In this expression, however, the influence of temperature and the threshold effect of low phytoplankton concentration must be included.

4. *Should state variables for fish and benthos be included in a eutrophication model? How?*

Reported by P. Mauersberger

Theoretical considerations as well as recent limnological observations suggest that mass balances for fish populations should be included in a eutrophication model. It was, for example, observed in a 60-hectare reservoir in Czechoslovakia that the algal concentration was significantly reduced by a controlled low stock of predatory fish. This may even be used as a management tool in situations where it is impossible, or too expensive, to reduce the input of the limiting nutrient.

The effects of fish can sometimes be simulated in the eutrophication model simply by modifying the mortality coefficient in the zooplankton equation. But for a more detailed model it appears necessary to include fish explicitly as an additional state variable. In this case the balance equation(s) for the fish component(s) may be similar to those for other consumer groups. While the biomass for the fish component(s) may be high, the energy or mass fluxes between the higher trophic levels are small. It may be reasonable to distinguish in some instances between juvenile and adult parts of the fish population and to take time variations into consideration. Of course, it is not easy to get the basic information about fish biomass, which is required when fish are included as a state variable in a eutrophication model.

The complete model of a shallow lake must in most cases include benthic components, and also exchange processes between sediment and the water body, in order to obtain the correct nutrient balances. In many eutrophic lakes the primary production by the phytobenthos is small in comparison with the primary production by the phytoplankton. Therefore, we can assume that only the upper nonshaded parts of the submerged plants assimilate nutrients. Thus the growth of phytobenthos does not depend upon their biomass, but upon the optical properties of the overlying water body. Some observations suggest that the decrease of phytobenthos in eutrophic lakes is combined with an increase of phytoplankton.

5. *How should nutrient exchange between sediment and water be modeled? How much detail would be needed in the description of biological, chemical, and physical processes taking place within the bottom sediment? Should the description be different for aerobic and anaerobic conditions?*

Reported by M.W. Lorenzen

The exchange of nutrients between sediment and water is a key process in the simulation of eutrophication. Unfortunately, available knowledge shows that the rate of exchange varies considerably from lake to lake and is a function of time.

A model that includes a "reactive" sediment layer, oxidized and reduced conditions, sorbed phosphorus, chemically fixed phosphorus, dissolved phosphorus in interstitial waters, and iron compounds was presented. The processes considered in this model are sedimentation and decay of organic material; desorption, reduction, and oxidation of iron; adsorption of phosphorus to iron compounds; chemical reactions and different redox conditions; transport between water and sediment and transport into the deeper sediments; and oxygen transport. All the reactions were of first order, and the adsorption followed Langmuir's isotherm.

In general it was apparent that sediment/water exchange processes should be considered in more detail. However, there has not yet been sufficient field experience to determine the most appropriate formulations.

6. *In the exchange process description, which parameters are general, and which are specific? Is it possible to establish a relationship between these parameters and some sediment characteristics? How are laboratory experiments with substracted (disturbed or undisturbed) bottom-sediment samples relevant to modeling the exchange of nutrients?*

Reported by G. van Straten

From the discussion in conjunction with question A5 it is clear that some basic models have been developed for the sediment subsystem, but they have not yet been examined sufficiently in relation to available data. Therefore, the alternative approach is to conduct laboratory experiments, preferably with undisturbed sediment samples. By varying the pH and oxygen and nutrient content in the overlying water, it should be possible to provide the information needed to derive parameters for a schematic submodel. An example of such a submodel was presented. In this submodel only interstitial phosphorus and exchangeable and nonexchangeable forms of phosphorus in the solid phase are represented. The nonexchangeable portion is determined from the phosphorus content in older sections of the sediment, using the argument that this type of phosphorus must be nonexchangeable, since it is present in the sediment. Such an argument is questionable, for it is known that bioturbation may be responsible for mixing the sediment up to a depth of 10-15 centimeters.

The submodel presented gives an indication of the problems that may arise from such an empirical approach. Since the simulation results of the submodel were not satisfactory, it was necessary to expand the model by adding other phosphorus components, thereby introducing more parameters. It may be concluded that although empirical models are of value, the final solution can only come from a better understanding of the real chemical/ biological behavior of the sediment.

In the case of simultaneous diffusion and chemical reaction, which is a characteristic feature of undisturbed sediment layers, some insight may be obtained from a classical chemical engineering approach. The analysis of chemical engineering systems began in the 1950s. The enhancement of material flux toward or from the sediment due to reaction can be characterized by two dimensionless parameters, the Hatta number and the infinite enhancement rate, which can both be derived from measurable properties of the sediment. Such a theory explains the observed relationship of oxygen uptake rates according to the square root of the oxygen concentration in the overlying water. It also demonstrates the decrease in uptake or release rates as time proceeds, which may be of significance in the interpretation of laboratory experiments for parameter estimation.

7. *How can the settling of phytoplankton and detritus be represented in the model?*

Reported by J. Fischer

Vertical transport results from a settling velocity of about 5-10 m/day and a vertical mixing. When considering the problem of a reduction in the amount of phytoplankton by diffusion, one has to suppose for the velocity that $V = f(D, \text{physical state})$, D = diffusion coefficient, in order to avoid contradiction.

Some investigations in the Netherlands have shown that there is a better correlation between settling rate and light than between settling rate and wind speed. It is probable that density gradients play an important role. Hypolimnion diffusion is much greater than epilimnion diffusion.

Since algae can change density, it might be necessary to represent this factor in the model, in order to get an accurate description of the settling rate. It is certain that the physiological condition of the algae plays an important role in determining this rate; typically the settling rate of the phytoplankton is more rapid when their concentration is decreasing. However, it is still uncertain whether such settling velocities have been measured realistically. Settling velocity does not appear to be a fundamental physical quantity, and there is little confidence in trap methods, although a rough estimate can be obtained by this kind of direct measurement.

8. *How do we model the following chain of processes: organic nitrogen → ammonium → nitrite → nitrate? Also how can stable dissolved organic nitrogen such as humus be modeled?*

Reported by L. Lijklema

Since phytoplankton show little preference for either ammonia or nitrate as a nutrient, it was considered unnecessary to give a detailed description of the nitrogen conversions. In most ecological models the process is represented as organic nitrogen → ammonium → nitrite → nitrate, unless substantial oxygen consumption is involved in those conversion reactions. Also in many situations nitrate is the predominant nutrient in the water.

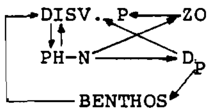
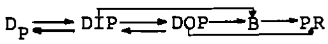
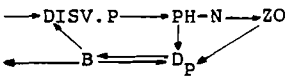
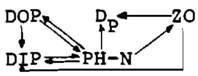
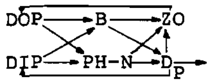
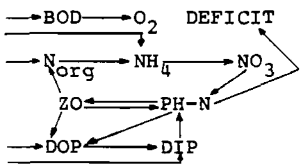
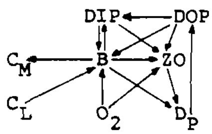
In chemostat experiments strongly fluctuating concentrations of nitrifying organisms may occur, but in the field such variations are generally slow. Therefore a temperature-dependent first-order conversion process for available nitrogen will be sufficiently accurate for most practical purposes.

9. *How do we model the organic → ortho-phosphorus process?*

Reported by V.J. Bierman, Jr.

It was recognized that the transformation of phosphorus in the natural environment is the result of many complex and simultaneous chemical and biological processes. The whole process can be simulated by using very detailed multicompartment kinetic models. In practice, the choice of the approach will depend on the particular objectives and the available data. Table 1 includes a repr̀esentative sample of various approaches that have been proposed for modeling phosphorus transformations.

Table 1. Some approaches to the modeling of phosphorus transformations. [a]

NN	Schemes	Authors
1	DOP → DIP	Skopintsev (1938) Maksimova (1972)
2	DIP → B → DIP	Grill and Richards (1964)
3	DIP ⇄ PP ⇄ DOP	Watt and Hayes (1963)
4		Corner (1973)
5		Barsdate et al. (1974)
6		Porter et al. (1975)
7		Richey et al. (1975)
8		Richey (1977)
9		Thomann et al. (1973)
10		Aijzatullin and Leonov (1975)

[a] DOP, DIP - dissolved organic and inorganic phosphorus
DISV.P - sum of dissolved organic and inorganic phosphorus
PP - sum of particulate phosphorus
D_P - detrite phosphorus
B - bacterial phosphorus
PR - protozoan phosphorus
PH-N - phytoplankton phosphorus
ZO - zooplankton phosphorus
BOD - biochemical oxygen demand
O_2 - oxygen
N_{org} - organic nitrogen
NH_4 - ammonium nitrogen
NO_2 - nitrite nitrogen
NO_3 - nitrate nitrogen
C_L - labile carbonous organic matter
C_M - organic matter transformed by bacteria

Recent experimental evidence has emphasized the importance of protozoa and zooplankton types in the transformation of organic phosphorus. It was also reported that the bacterial uptake of inorganic phosphorus may increase the bacterial consumption of organic phosphorus under certain conditions, and that the presence of many predators in proportion to bacteria can greatly accelerate the bacterial transformation of organic phosphorus.

10. *How can algal succession be represented in the model? As blue-green algal blooms are an important problem, how could we at least distinguish between "other algae" and blue-green algae?*

Reported by V.J. Bierman, Jr.

Whether or not algal succession should be represented in a model depends upon the particular circumstances of the system being considered. In Lake Balaton, for example, it was reported that in the summer nongrazed Pyrrhophyta are becoming predominant over other grazed species, and that accurate results cannot be obtained for zooplankton concentrations by using only a single algal group in the model. It is suggested that at least five phytoplankton types should be included in the Lake Balaton model: Bacillariophyceae; Pyrrhophyta, Cyanophyta, Chlorophyta, and others.

Another aspect of algal succession was discussed, although it has no direct relationship to algal production rates. In Lake Balaton, a substantial increase in blue-green algae from the sediment layer, where they grow in large quantities, was reported. The opinion was expressed that ecological models will be unable to describe this phenomenon adequately.

In an attempt to describe the buoyancy phenomenon in another lake, it was reported that settling velocities of 1 m/day to 20 m/day were required in order for them to correspond to the data. In this case the total blue-green biomass in the first 2 centimeters of sedimented material was approximately the same as the total blue-green biomass in the water column.

There was a suggestion that, in general, variables for algal succession should be included in the model when

-- Silica is limiting--this will cause a succession in species from diatoms to nondiatoms
-- There is a substantial blue-green bloom--these species have different characteristics from other types
-- Fixation of nitrogen is shown to take place--this frequently occurs in lakes, where the phosphorus/nitrogen ratio is relatively low

Criteria for selecting algal species with functional characteristics best suited to a given set of circumstances were presented using a self-optimizing (self-organizing) principle with maximization of a specified goal function for the algae. For complete information see Radtke and Straškraba (1977).

The temperature preferences of blue-green algae were discussed. Blue-greens are usually observed to be dominant when the water temperature exceeds 20 °C. However, it was pointed out that there are certain blue-green species, for example, *Oscillatoria*, that grow well at lower temperatures.

It was pointed out that the microclimate around blue-green algae may lead to certain functional differences as compared to other types of algae. Blue-greens are known to maintain a close relationship with symbiotic bacteria. This characteristic may be responsible for influencing the pH value and uptake kinetics for phosphorus and carbon dioxide.

Reference was made to results achieved by Shapiro, which indicate that blue-greens are more efficient than other species for carbon dioxide uptake. This conclusion was challenged because the experiments performed did not measure independently the effect of the change in pH on the carbon and the phosphorus-uptake kinetic. The distribution of the dissolved forms of both nutrients changes as a function of pH.

There was discussion of the point that commonly accepted notions of blue-green settling and their freedom from grazing pressures may not be correct in all cases. Apparent settling velocities for blue-green algae may not be slower than settling velocities for other species, but may be more rapid due to clumping and colony formation. Recent experimental work by McNaught on Lake Huron has indicated that blue-green species may actually be preferred by the zooplankton when compared to certain non-blue-green species. In these cases it is not clear whether the zooplankton assimilate the blue-greens or merely ingest them.

Local blue-green algae problems are not primarily caused by their rapid mass growth, but rather by their buoyant rising to the surface, where they form clumps. Under these circumstances the well-known disadvantage of biomass decay that is highly concentrated in a relatively small part of the lake will appear. This phenomenon is very difficult to include in the model.

11. *Do we need to include anaerobic conditions, denitrification, and nitrogen fixation in the model, and, if so, how?*

Reported by G. van Straten

One approach in which the effects of nitrogen fixation upon the nitrogen budget can be taken into account is to set the rate of nitrogen fixation proportional to the nitrogen deficiency, calculated from the discrepancy between the soluble nitrogen and phosphorus ratio in the water and the phosphorus/nitrogen uptake rate. There seems to be no sharp threshold concentration for nitrogen fixation--some 300 μg nitrogen/liter of soluble nitrogen has been mentioned as a suitable figure. A simple solution is to set the fixation rate at zero for 300 μg nitrogen/liter and to account for a linear increase if soluble nitrogen drops to zero. This is a more elaborate case of the simple approach whereby nitrogen is not allowed to become limiting to the nitrogen-fixing blue-green algae; one merely traces the changes of nitrogen during their growth. To prevent problems regarding the ratio of nitrogen-fixing to non-nitrogen-fixing algae, the introduction of a separate state variable for nitrogen-fixing algae might be envisaged.

There was general agreement about the occurrence of denitrification in the sediment. The theoretical possibility of denitrification in the oxygen depleted core of algae clumps does not seem to be important in practice. For denitrification to occur it does not seem to be of great significance that the sediment is entirely anaerobic, because there will always be nitrate present in the sediment in the anaerobic portions, since nitrate penetrates much deeper than oxygen. A simple first-order reaction has been applied as a reasonable approximation of denitrification, although the results of analysis of the simultaneous diffusion and reaction system would suggest a square root dependence (a half-order dependence). Of course, a more detailed model would require the incorporation of denitrifying bacteria.

12. *How do we select the necessary number of state variables for solving a specific problem? A greater number of state variables involves the introduction of more parameters, and more measurements must also be carried out, while fewer state variables may not describe the structure of the system in sufficient detail. How do we find the point of balance? What is the role of chemical data?*

Reported by M. Straškraba

Three methods were suggested for selecting the number of state variables to be included in the model. The first is to use the criterion of ecological buffer capacity β which is expressed as the ratio between change in loading and change in the state variables of interest--e.g., $\beta = (\Delta \text{ loading})/(\Delta \text{ PS})$, where PS is the state variable representing soluble phosphorus. For further details, see Jørgensen and Meyer (1977). It was stated that β cannot be used for comparing several models. It is noticeable that β is time dependent, if such external factors as temperature and irradiance are functions of time.

The second method is one to decide which degree of model complexity is sufficient for a specific purpose. The variance of the property that the model is intended to predict is affected by the number of state variables or parameters. We assume that the purpose of the model is to predict the consequences of some changes to the system under study, such as increased loading. We also assume that the change can be quantified by some numerical quantity P (such as the maximum concentration of phytoplankton). A model based on N_1 parameters, α_{N_1}, will yield an estimate $\hat{P}(N_1)$ of P, and this estimate will have a sampling distribution, because its parameters are calculated from observed data that can be regarded as random variables. Similarly, a more complex model with N_2 parameters α_{N_2} $(N_2 > N_1)$ will yield a second estimate $\hat{P}(N_2)$ of P; the closer $\hat{P}(N_1)$ is to P, the better the model. It is suggested that the root mean square error could be taken as a measure of the model's "goodness-of-fit." However, P is never known, therefore some less satisfactory alternative has to be used, such as $\mathrm{var}\{\hat{P}(N_i)\}$. Now $\mathrm{var}\{\hat{P}(N_i)\}$ is a function of the sampling variances--i.e.,

$$\mathrm{var}\{\hat{P}(N_i)\} = \sum_{j=1}^{N_i} \left(\frac{\partial \hat{P}}{\partial \alpha_j}\right)^2 \mathrm{var}\{\hat{\alpha}_j\} + 2 \sum_{j=1}^{N_i} \sum_{k=1}^{N_i} \left(\frac{\partial \hat{P}}{\partial \alpha_j}\right) \left(\frac{\partial \hat{P}}{\partial \alpha_k}\right) \mathrm{var}\{(\hat{\alpha}_j, \hat{\alpha}_k)\} \quad ; \quad j \le k \quad . \tag{6}$$

This quantity will have more or fewer terms depending upon whether the model is complex (N_1 large) or simple (N_i small). We could therefore plot $\mathrm{var}\{\hat{P}(N_i)\}$ as ordinate against N_i as abscissa, and choose that value of N_i, beyond which an increase in the number of parameters gives little or no reduction in $\mathrm{var}\{\hat{P}(N_i)\}$. This approach could be adapted to include consideration of the greater cost associated with more complex models; if the cost were taken as a linear function of N_i, and A and B are constants, the number of parameters would show that

$$\mathrm{Cost} = A + BN_i \quad ,$$

and then the picture given in Figure 2 would be obtained. The model with N* parameters would then be adequate for the specified purpose.

The third method has not yet been tested in practice. It involves the development of some statistical indices similar to the indices of diversity, where, instead of the number of species, the number of variables would be used.

Generally it was felt that we need more systems analysis methods, in order to choose the correct level of complexity, and to validate the more complicated models. Other aspects of validation were raised, particularly the impossibility in most

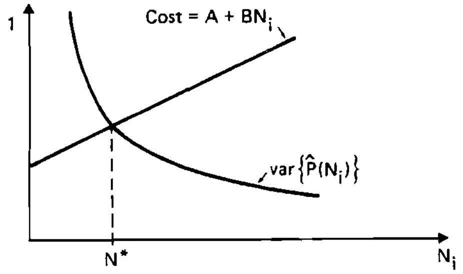


Figure 2. Curves showing the accuracy, var $\{\hat{P}(N_i)\}$, and cost associated with having a model with N_i parameters.

instances of validating the model under different conditions of nutrient loading. Even when loading are changed, the response in most cases will be delayed; thus, it will be necessary for several years to elapse before an accurate validation can be undertaken. It was suggested that a validation of the same model should be applied to a number of lakes with different loadings. Finally, it may be noted that the level of model error increases with an increase in the number of parameters.

13. *How reliable are present-day ecological models? To what extent is it possible to predict the response of the lake to changed loading?*

Reported by P. Mauersberger and K. Bauer

From the mathematical point of view the equations characterizing a microscopic, deterministic, water quality model form a system of nonlinear differential equations with boundary conditions and initial values. It is difficult to investigate the existence and uniqueness of the solution to these equations. From the thermodynamic point of view the aquatic ecosystem is a nonlinear open system, exchanging energy and matter with the environment, which is far from thermodynamic equilibrium. In completing this system of basic equations by adding entropy, it becomes possible to apply the methods and results of the modern theory of thermodynamics of irreversible processes (Prigogine, 1972) to aquatic ecosystems. This theory offers local and global evolution and stability criteria. The mutual effect of entropy-producing and entropy-reducing processes inside the water body and across its boundaries regulates the structure, state, and further development of the aquatic ecosystem. Since the nonlinear basic equations are shown, in general, to have more than one solution when the system is sufficiently distant from thermodynamic equilibrium, fluctuations (generated by the system itself or excited by external factors) play an important role during the transition of the ecosystem to a new structure. The succession of structures characterizes the anthropogenically influenced processes of self-adaptation and self-organization of the ecosystem. This theory has not yet been fully elaborated. In particular, the regulating mechanisms of living subsystems should be included; thermodynamic and cybernetic considerations must be combined, and stochastic elements should be incorporated into the model. Despite its incompleteness, such a theory may provide precisely the "comprehensive" (or "condensed") approach to water quality modeling that was called for in the introduction to this workshop.

Many problems relating to the predictability of the model's responses to changed loading remain open to discussion. For example, is the influence of microcontaminants, molybdenum oxide, glycine, and the like taken into account in such a way that the model reacts correctly to changes in the concentrations of these substances.

14. *Is it possible to develop models for predicting the rate of eutrophication? Should simple or complex models be used? What is the potentiality and what are the limitations of the models? Are the analytical expressions of the relationship between phytoplankton biomass and total loading of nutrients, average depth, retention time of water, and the like valid (Vollenweider index)?*

Reported by V.A. Vavilin

If a modification of Vollenweider's approach is used, then

$$dB/dt = \frac{1}{T}B + \mu(X,t,I)B \quad ,$$

$$dX/dt = \frac{1}{T}(X_o - X) - \gamma\mu(X,t,I)B \quad , \qquad (7)$$

where B is the phytoplankton concentration, X is the limiting nutrient concentration, $T = V/q$ is the mean retention time (V is the volume of the water body, q is the flow), γ is a stoichiometric coefficient, $\mu(X,t,I)$ is the phytoplankton growth rate as a function of the limiting nutrient concentration X, (t) is temperature, I is the light intensity, and X_o is the nutrient concentration of the influent.

Let us now assume that

$$\mu = \mu_m(t) f(I) \phi(X) \quad , \tag{8}$$

where

$$f(I) = \begin{cases} 1 & I > I_m \\ I/I_m & I < I_m \end{cases} \quad , \tag{9}$$

and where I_m is a saturation constant. Then

$$\phi(X) = \frac{X}{K_x + X} \quad , \tag{10}$$

in which K_x is a half-saturation coefficient.

The intensity of light at depth h is

$$I_h = I_o \cdot e^{-(n_o+nB)h} \quad , \tag{11}$$

where I_o is the light intensity at the surface, n_o is the light-absorption coefficient for water, n is the specific extinction-absorption coefficient for phytoplankton. Thus, if light intensity is a limiting factor at depth

$$h_m = \frac{1}{n_o + nB} \ln \frac{I_o}{I_m} \quad , \tag{12}$$

then

$$I_m \leq I_o \leq I_m e^{-(n_o+nB)H} \quad , \tag{13}$$

where H is the maximum depth.

The mean value $\bar{\mu}$ is given by

$$\bar{\mu} = \frac{1}{H}\left(\int_0^{h_m} \mu dh + \int_{h_m}^{H} \mu dn\right) - \mu_m(t)\frac{1}{(n_o + nB)H}$$

$$\times\left(1 + \ln\frac{I_o}{I_m} - \frac{I_o}{I_m} \cdot e^{-(n_o+nB)H}\right)\frac{X}{K_x + X} \quad , \tag{14}$$

and, instead of the influent concentration of limiting nutrient X_o, let us now consider the mean nutrient loading X_ε

$$X_\varepsilon = \frac{X_o \cdot q}{S} \quad , \tag{15}$$

in which S is a surface of water body.

Under a steady-state condition these equations are transformed into

$$\begin{cases} -\frac{1}{T} + \tilde{\mu} = 0 \\ \frac{1}{T}(\frac{X_\varepsilon \cdot S}{q} - X) - \gamma\bar{\mu}B = 0 \quad . \end{cases} \tag{16}$$

In the linear case $K_x >> X$. If

$$1 + \ln\frac{I_o}{I_m} >> \frac{I_o}{I_m} \cdot e^{-(n_o+nB)H} \quad , \tag{17}$$

then

$$B \approx \frac{\frac{X_\varepsilon \cdot S}{qK_x} \cdot \alpha - \frac{n_o}{n}}{1 + \gamma\frac{1}{K_x} \cdot \alpha} \quad , \tag{18}$$

where

$$\alpha = \frac{\mu_m(t) \cdot T}{nH}(1 + \ln\frac{I_o}{I_m}) \quad . \tag{19}$$

From the Equation (18) it is easy to obtain

$$X_\varepsilon = \frac{\gamma \cdot q}{s} \cdot B + \frac{q}{s} K_x \left(B + \frac{n_o}{n}\right)\frac{1}{\alpha} \quad . \tag{20}$$

We can distinguish between

$B < B^{+}$ --oligotrophic state

$B^{+} < B < B^{++}$--mesotrophic state

$B > B^{++}$ --eutrophic state

From this Figure 3 is obtained, where the parameters γ, q/s, n_o/n are constants in a specified case.

In a more general case when Equation (13) is valid, the following expression can be obtained.

$$X_\varepsilon = \frac{\gamma \cdot q}{\delta} \cdot B + \frac{q}{\delta} K_x \frac{1/\alpha}{\dfrac{1}{n_{o/n} + B} - \dfrac{1}{\alpha}} \quad . \tag{21}$$

Equation (21) is plotted in Figure 4.

It is possible to consider some modifications of Equation (7). As given, the model permits an accurate estimation of the trophic state, when the limiting-nutrient loading is changed. The classification of lakes, according to deduction from the Vollenweider-type diagram, is not suitable in some instances.

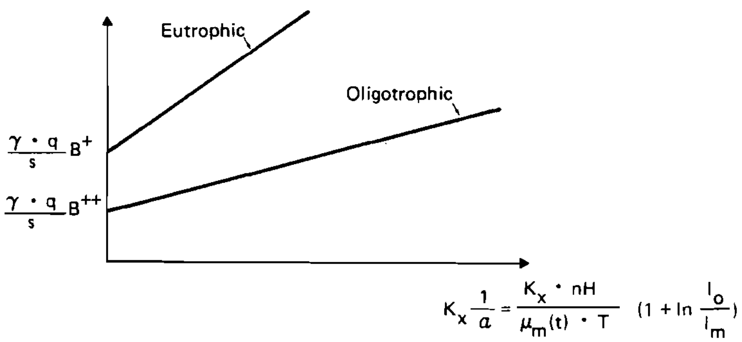


Figure 3. The distinction between oligotrophic, mesotrophic, and eutrophic states on the basis of Equation (20).

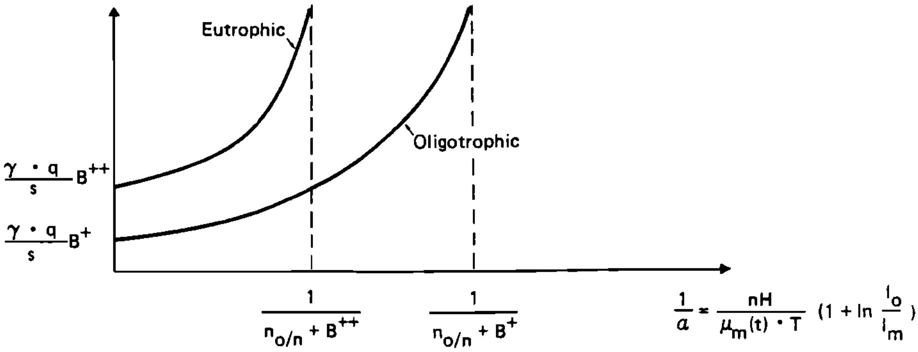


Figure 4. The distinction between oligotrophic, mesotrophic, and eutrophic states on the basis of Equation (21).

Dillon improved this approach, by considering the dilution rate as an important parameter.

The approach presented above depends upon retention time, depth and intensity of light, maximum growth rate, etc. The important question is whether the steady-state approach is valid both for the model and for the real lake system. In systems analysis it is important to establish whether the local stability conditions are indicative of globally stable dynamic behavior. The validity of the steady-state approach depends primarily upon seasonal variations of nutrient loading.

Vollenweider's approach is attractive because of its simplicity--it requires no large-scale experimental work. However, from a decision-making point of view, it seems necessary to construct and investigate dynamic models.

15. *What parameters and processes should be considered for the lake subject to acidic deposition?*

Reported by M.W. Lorenzen

A survey of lake susceptibility to acid precipitation would be useful.

It was noted that literature from Norway and Sweden has reported on studies in those countries; and the Workshop offered no final conclusion about how to solve this problem.

16. *A suggestion for an IIASA publication: should IIASA publish a list of parameters?*

Reported by L.J. Hajdu

The value of model parameters is of great interest for all scientists working in the field of ecological modeling. Many data are to be found in the literature, but a handbook containing the most important parameters, such as half-saturation constants, death, sinking, grazing rates, minimum phosphorus concentrations in algae, would be useful.

Concurrent with the development of the field, revision of these parameters values must take place. It was recommended that a list of parameters be completed according to standard units (SI), with the necessary conversion tables included. It might also be of interest to standardize techniques of parameter measurement--e.g., for the phosphorus half-saturation constant, since it is not a constant parameter but changes according to ecological and hydrological factors. The methodology of parameter measurement may appear to be of marginal interest to IIASA, but undoubtedly the reality of a model is to a great extent dependent upon the reliability of the basic parameters.

Modern limnology textbooks reflect this attitude to modeling, but they may be of limited value to the practical model builder, since they usually contain few suitable model parameters. The general validity of these parameters is not fully understood, but if a collection of widely distributed literature data were available, it would make possible the study of the generality of the parameter values suggested by several authors. It would also be useful to include a recommended literature list and the addresses of authors and contributors in the handbook suggested above.

It was announced that such a handbook was being prepared in Denmark and would be ready for publication in the autumn of 1978. The data would be available on tapes. Approximately 50 international journals and 500 books have been reviewed, and the handbook will probably cover 75 percent or more of the total relevant literature. Parameter values only are compiled without critical evaluation, but all data are accompanied by appropriate references and the specific measuring conditions. In this handbook no average values or standard deviations are calculated; the reader should evaluate the reliability of the parameter values for himself. An Environmental Protection Agency (EPA) report containing a review of equations and expressions used in water quality models will soon be ready in the United States. The report will also include a list of parameter values, although the review of the literature is limited.

It was pointed out that confusion may arise in the comparison of different parameter values, such as half-saturation constants, since they might have different physiological implications from one case to another.

17. *How and when should quick phosphorus recycling due to phosphate excretion by phytoplankton be considered in the model?*

Reported by F. Chahuneau

In the literature inverse relationships between alkaline phosphatase activity and phosphorus cell concentration have often been reported. It is not well established whether the enzymes are located at the cell surface, or whether they are released into the environment. Under the first assumption the algae will behave exactly as if they were able to assimilate dissolved organic phosphorus directly--a factor that is not usually taken into account in models. However, the rate of dissolved organic phosphorus hydrolysis is at least one order of magnitude lower than the average phosphorus uptake rate. This leads to the conclusion that this phenomenon can usually be disregarded in the model.

B. HYDROLOGICAL TOPICS

1. *Under what conditions is it necessary to model the hydrodynamics of the lake circulation (equations of motion, wind stress, eddy diffusivity, Coriolis effects)?*

Reported by M. Markofsky

No definitive answer can be given to the question raised. In general, most of the comments suggested that the degree to which the hydrodynamics should be modeled depends upon the problem being investigated and upon the time scales of the relevant phenomena.

Two case studies were presented. The first required the prediction of the likelihood of entrainment by a 3,200-megawatt power station in the western basin of Lake Erie, where the mean depth is approximately 6 meters. Here a three-dimensional description of the wind-induced velocity field was found necessary for use as input to the transport equation describing larval transport (Lick et al., 1976).

In the second case study, which concerned a drinking water intake on Saginaw Bay, it was felt that the primary current patterns could be sufficiently well deduced by intuition, so that no mathematical description of the wind-induced circulation was required. In this example it had been observed that strong odors were associated with a rapid increase in intake temperature. The increase in intake temperature was judged to be associated with a change of wind direction, as shown in Figure 5. Since the intake is near the bottom, cooler lake water is withdrawn for the situation represented in Figure 5a and warmer bay water for the situation represented in Figure 5b. The suggested solution to the water quality problem is to construct an addition to the intake, in order to allow the intake to draw surface water during the wind condition of Figure 5b.

It was proposed that the degree of hydrodynamic description depends upon the relative time scales of the biological and hydrodynamic processes. For example, if the biological processes occur on a time scale considerably larger than the circulation and wind-induced mixing time of the water body, the details of the hydrodynamics can probably be neglected.

In any case, hydrodynamic and water quality models are very often solved in quite different time steps, Δt, and space steps, Δx, since Δt and Δx for the hydrodynamic resolution are generally considerably smaller than those required for the water quality description. This is increasingly important when considering an ecosystem description in many state variables.

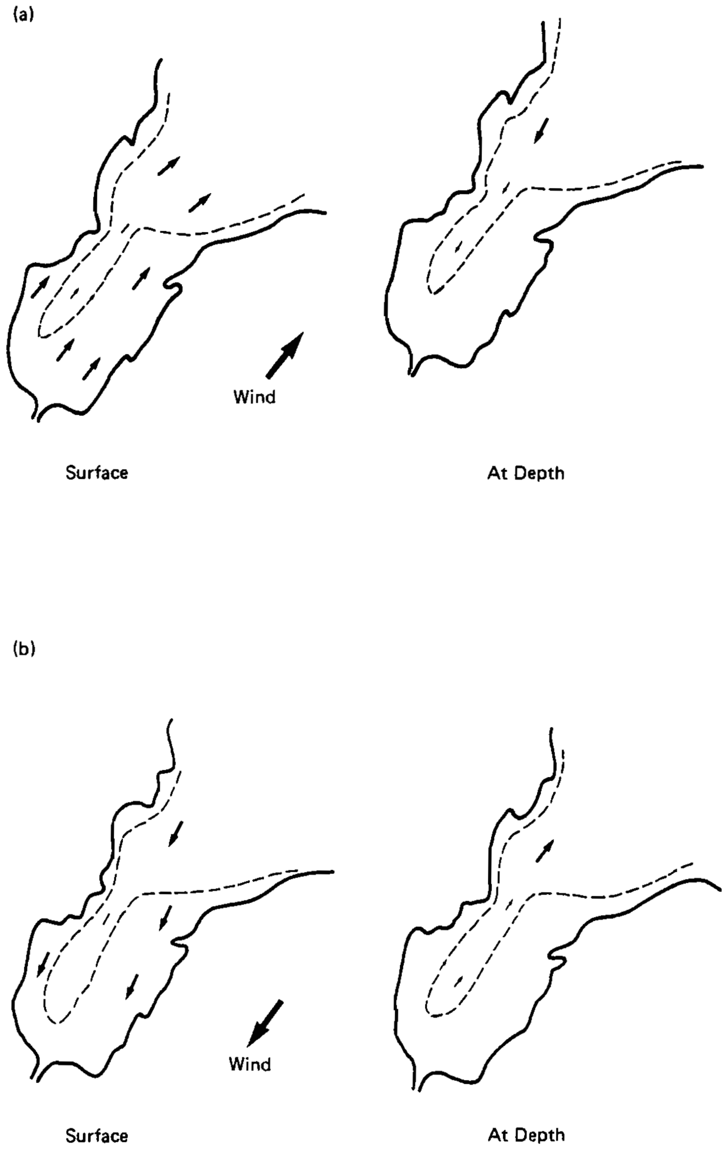


Figure 5. Wind-induced circulation in Saginaw Bay.

2. *What is the relative importance of lake or reservoir inflows and outflows in the vertical and horizontal exchange, as opposed to vertical and horizontal mixing by diffusion or wind?*

Reported by M. Markofsky

Case studies in reservoirs in which thermal stratification occurs were reported. The varying input elevation of the inflowing water is a function of both the inflow temperature and the thermal structure of the reservoir. Disregarding the effect of vertical advection in the model results in vertical dispersion coefficients that are a function of depth and time.

A sensitivity analysis for a reservoir with a retention time of 1 year was presented. The governing equation for temperature prediction is

$$\frac{\partial T}{\partial t} + V\frac{\partial T}{\partial y} = \frac{1}{A(y)}\frac{\partial}{\partial y}\left[AE\frac{\partial T}{\partial Y}\right] + q_i\frac{(T_{in} - T)}{A} - \frac{1}{A\rho C_p}\frac{\partial(\phi_b A)}{\partial y}\quad , \quad (22)$$

and for mass ($j = 1 \rightarrow n$)

$$\frac{\partial C_j}{\partial t} + V\frac{\partial C_j}{\partial y} = \frac{1}{A(y)}\frac{\partial}{\partial y}\left[AE_M\frac{\partial C_j}{\partial y}\right] + q_i\frac{C_{inj} - C_j}{A} + \frac{\text{sources}_j}{\rho} - \frac{\text{sinks}_j}{\rho}\quad , \quad (23)$$

T is the temperature; y is the depth; V is the vertical velocity (a function of depth and time); E is the dispersion coefficient for temperature; E_M is the dispersion coefficient for heat; A is the horizontal cross sectional area; q_i is the inflow rate/height; ϕ_b is the solar radiation penetrating the surface; T_{in} is the inflow temperature; and C_{in} is the inflow concentration.

The vertical velocity field, v(y,t), may be calculated from the inflow and outflow rates and density considerations. Varying E_M from 0 to 100 times the molecular value produces no changes in the concentration profiles; this indicates that for a reservoir of the above size, or retention time, an accurate description of the velocity field that allows a dispersion term to be disregarded is assumed.

For shallow reservoirs it is to be expected that wind mixing plays a dominant role in the vertical and horizontal exchange processes. The shorter the residence time the greater the role of inflow and outflow on the horizontal exchange phenomenon. However, the better the mathematical description of the flow and wind-induced current fields, the lower is the reliability of the dispersion coefficient as a fitting parameter. The extent to which one attempts a detailed formulation of the velocity field, however, is related to the discussion associated with question B1.

3. *What are the factors that cause a concentration of suspended sediment in lakes? How do we model the resuspension of bottom sediment?*

Reported by G. van Straten

Turbulence in the region of the sediment bed is the main factor that determines whether there will be a resuspension of sediment particles from the lake bottom. Turbulence may originate from two possible sources. The first is the shear at the bottom surface of the lake caused by a return flow along the bottom in the case of wind-driven circulation or by a transition flow in a narrow part of the lake. The effects are quite similar to those of sediment transport in rivers, although the range of velocities that might occur is less. Turbulence can also be generated by waves, especially by breaking waves.

The time during which a particle remains in suspension might be assumed to be a governing factor in the mathematical modeling of ecosystems. This point of view was implicit in the reference to the "saltation" theory of resuspension, whereby particles are dispersed upwards, and after some time settle again. It is also necessary to know the age distribution of particles in suspension; but effects such as cohesion forces--due to potential differences in the electrochemical properties of the sediment flocs--present difficulties in determining the age distribution.

Another important consideration is that in many cases there exists no sharp interface between water and sediment. This leads to the concept of a "liquid sediment," the formation of which can be analogous to the erosion of the thermocline due to waves induced by wind.

A Scottish lake was given as a practical example. In this case a striking coincidence was observed between the change in sediment composition and the line where the wave length, caused by wind, reached a depth equal to half its length.

In a Dutch shallow lake, turbidity seemed to bear more relation to inlet water solids concentration than to resuspension. Although during a severe storm some resuspension was to be expected in the shallower part of the lake, almost no increase in turbidity was observed. A possible explanation might be that the deeper parts of the lake acted as an effective "dust-binder," since quieter conditions may be expected there, thus allowing sediment to be deposited.

C. GENERAL TOPICS

1. *How should the time-dependent meteorological factors be included in lake models?*

Reported by J.F. Paul

The following question was raised: How is it possible to calculate hydrodynamic currents for dominant wind conditions (steady-state currents)? It was answered as follows. The hydrodynamic results should be processed so that they can be used in ecological models. For such models the speed and direction of the wind should be chosen in some stochastic manner that is changed at every time step. Over a time scale of weeks, which is in most cases the smallest time scale of interest in ecological models, the wind pattern can then be considered as this steady wind state. For prediction, it is necessary to look at historical wind patterns, in order to obtain a stochastic model of wind behavior. Major events will have occurred at different times and in different numbers, but over several years these fluctuations may be averaged to give typical wind conditions. For calibration and validation, wind data for a specific year should be used; thus, particular events, which could produce major effects, will be included in the calibration and validation. The above suggestions would seem to be an efficient way of coupling hydrodynamic and ecological models in order to make hydrodynamic calculations once only for each lake.

Generally, three approaches taking meteorological factors into consideration in lake models were suggested:

-- Time-varying inputs of wind or other external factors are used in the hydrodynamic model.
-- A correlation analysis between meteorological data and influenced variables or parameters is carried out.
-- A correlation analysis between calculated or estimated variables and parameters and meteorological data is carried out.

It was stated that the method of averaging wind over time to provide an input forcing function may cause a strong interaction with the response of the system. In a network of small lakes connected by canals, a detailed description of the wind function was reported to cause a high frequency variation in output flows and water level, because of the interaction of the wind function with natural frequencies of the system. By averaging wind data over a longer period, these high frequencies can be filtered out, and the more important low frequency responses associated with the major flows are shown more clearly. When the wind velocity is averaged, however, one should remember that it appears in a squared form in the equation of motion.

2. *How do we distinguish between point and nonpoint sources, and how should they be considered in the model?*

Reported by D.O. Logofet

This problem is complex and it is probable that all nonpoint sources of nutrients, such as land runoff, urban runoff, precipitation, surface infiltration of ground waters, should be treated in a stochastic manner. One way of solving the problem is to carry out a correlation analysis between nonpoint sources and the observed behavior of the lake. But if the correlation coefficient appears to be too small, should we then conclude that these features are not correlated? It was mentioned, for example, that the correlation between the phosphorus concentration in Lake Balaton and the entire discharge into this lake appeared to be rather small, $r = 0.007$, although we know that there must be a correlation between these two variables. However, the correlation might be stronger if the land runoff, the flow input, and the rainfall are considered separately. It might also be stronger if the discharge is measured directly at the source point; but this approach requires careful treatment of each source, and adds complications to the model.

Another question related to this problem was raised. What values of time lags should be selected when a correlation analysis is considered? This is not easy to answer, because of the time variability of loading in different sources. Under all circumstances it seems necessary to take into account in the model the distribution pattern of nonpoint sources. To do so it is necessary to conduct measurements and exercises with hydrological submodels.

3. *Which parameter estimation procedure should be applied?*

Reported by M.B. Beck

One of the suggested procedures can be summarized as follows: since the calibration problem for a system with, let us say, 10-15 parameters to be estimated can be likened to the situation of "a blind man searching for a hole in a 10-15-dimensional room," it is necessary to limit the problem, in order to obtain tractable solutions. Suitable maximum and minimum bounds can be imposed on the permissible values of the parameters, where these bounds are derived from parameter estimates previously published in the literature. An automatic calibration procedure can, however, be carried out for making fine adjustments to the accuracy of a limited number of parameters (up to five), but reasonable *a priori* estimates of these parameters would be required. A third solution is to use the submodels containing relatively few parameters, which may be calibrated against data collected from an intensive field experiment--an experiment that might measure more variables at a higher sampling frequency than the routine monitoring program. The objective of this last option would be to calibrate those parts of the model that exhibit relatively rapid dynamic behavior, and that can be isolated, to some extent, from variations in the other parts of the lake's ecological system. There is also a possibility of examining individual process mechanisms by laboratory studies or by field studies in which it might be possible to isolate such process mechanisms.

The automatic calibration mentioned above is based on a sensitivity analysis, which provides a matrix of partial derivatives for the ratio of changes in the state-variable responses to changes in the parameter values. This matrix determines the directions of a search routine, which then attempts to locate a minimum value for the squared prediction error loss function.

Some improvement in the efficiency of this procedure might result from the use of Rosenbrock's search algorithms. It is important to notice that the automatic calibration procedure, as expected, cannot prevent some of the parameter values reaching and being constrained at their maximum or minimum permissible limits. In this context it must also be emphasized that several of the parameters may be time varying; thus during automatic calibration an average parameter value is found, and this does not take into account the possibility of seasonal variations.

Clearly, there are many difficulties in this area of ecological modeling, not the least of which relates to the credibility of the results; thus the following cautionary messages should be mentioned. Automatic calibration may be dismissed as a meaningless curve-fitting exercise; a predominantly hydrodynamic event might be erroneously calibrated within the ecological portion of the model; there are few cases where such a degree of detail is warranted by the overall objective of the modeling program.

4. *Is it possible to set up a more general procedure for handling a specific lake modeling problem? The procedure should focus on the method of selecting the process equations, on the number of observations that are required, and on calibrating and validating the model.*

Reported by M.B. Beck

Two presentations were given in response to this question. The first outlined a procedure for modeling

-- The definition of the problem (model application objectives)
-- The development of conceptual flow diagrams
-- The selection of the state variables--based on the preceding two steps and the concept of ecological buffer capacity (see also question A13)
-- The calibration of submodels against intensive sampling programs--notice that at this stage the model, or submodel, can be used as a scientific tool for evaluating the description that gives the best characterization of observed behavior
-- The fine adjustments to the total model against intensive and other available data
-- The validation of the model, preferably by reference to an independent set of measurements taken during a period with changed conditions (e.g., changed loadings)

The second presentation developed a more detailed discussion of the problem of model structure identification and parameter estimation. Both problems can be considered as parts of the overall calibration problem. This presentation deals with some topics that did not arise in the "Proceedings of the Workshop on Geophysical and Ecological Modelling of Deep Lakes and Reservoirs" (Jørgensen and Harleman, 1978).

It is worth noting that a number of parameter estimation procedures have been applied successfully for modeling purposes in other fields of study. Each estimation procedure can be approximately classified according to whether it is applicable to black box models or internally descriptive (mechanistic) models, and according to whether it is "off-line" or "recursive"--these terms will be defined below.

For black box models, the following procedures can be applied: least squares (R) (O), instrumental variable (R) (O), maximum likelihood (O), group method of data handling--a self-organization method.

For internally descriptive models, the maximum likelihood (O) procedure and the Extended Kalman Filter (R) can be applied. The symbol R denotes a recursive procedure, and the symbol O denotes an off-line procedure. Figure 6 is a pictorial representation of the differences between the two types of algorithm.

An off-line procedure keeps the parameter estimates constant at their *a priori* values, $\hat{\theta}^0$, while the complete block of time-series field data, from t_0 to t_n, is processed. A loss function, generally based on the errors between observed and model responses is computed, and an optimization algorithm for minimizing the loss function over the parameter space then computes updated parameter values, $\hat{\theta}^1$, for substitution in the next iteration through the data (i.e., from t_0 to t_n). It is this form of parameter estimation procedure that was discussed in response to question C3. A recursive parameter estimation algorithm, in contrast, computes updated parameter estimates, $\hat{\theta}^0(t_k)$, at each sampling instant, t_k, of the field data; the minimization of the error function is implicitly, rather than explicitly, included in the algorithms.

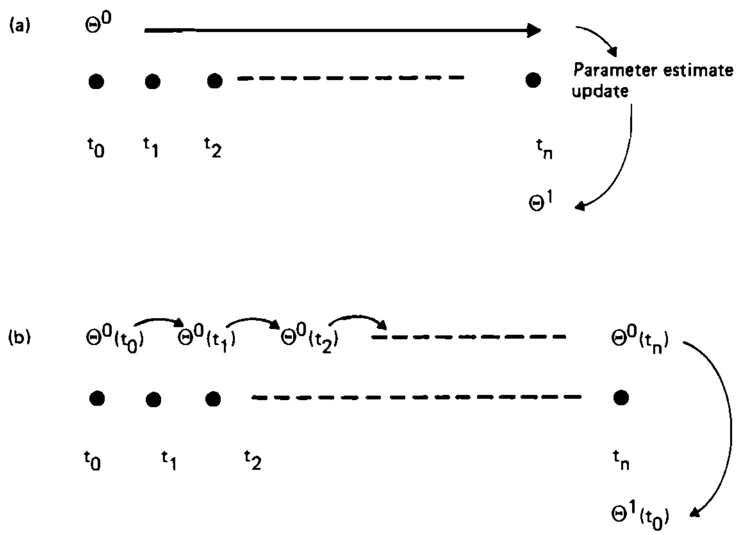


Figure 6. Methods of parameter estimation: (a) off-line and (b) recursive. Θ^0 represents the *a priori* values of the parameter estimates, and t_0 to t_n denotes the complete block of time-series field data.

At the end of the block of data, the estimates, $\hat{\theta}^0(t_n)$, are substituted for the *a priori* parameter values $\hat{\theta}^1(t_0)$ of the next iteration through the data. An example of each approach to parameter estimation can be found in Di Cola et al. (1976)--an off-line estimation scheme--and in Gnauck et al. (1976)--a recursive procedure.

One advantage of a recursive estimation algorithm is that it makes the estimation of time-varying parameter values possible--see Young (1974) and Young and Whitehead (1977) for a more complete discussion with special reference to problems of hydrological and water quality modeling. In fact, this particular aspect of recursive techniques lends itself to the solution of the model structure identification problem and may also lead to the same interpretation of the self-organization problem as that which was mentioned above by Straškraba. Figure 7 attempts to represent the essence of the model structure identification problem. Suppose that the state variables are represented by the nodes of Figure 7, and the parameter values represent "elastic" connections

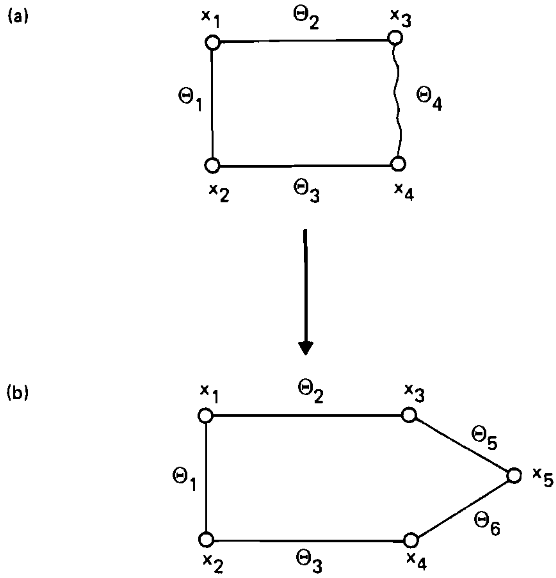


Figure 7. A conceptual picture of model structure identification. The nodes represent the state variables, and the connections between them denote the parameter values.

between the state variables--e.g., as in growth-rate and grazing-rate functions. If it is assumed that all the parameters have values that are constant with time, and yet the recursive algorithm yields an estimate for Θ_4, say, that is significantly time varying, one may question the correctness of the chosen model structure. The reasoning is as follows: the general purpose of the estimation procedure is to fit the state-variable predictions to the field observations; thus if there is any structural discrepancy between the model and "reality," this will be manifested as an adaptation of the model--i.e., the parameter values, towards "reality." Such time variations of the parameter values can occur for different reasons. For instance, the parameter may be truly time variant in some seasonal fashion, or there may be hidden problems of identifiability associated with the given model structure. For the purposes of our example in Figure 7a, however, we might suppose that the actual structure of the relationships underlying the observed behavior are better represented by the introduction of a new state variable and two new parameters, Figure 7b. These kinds of argument form the basis of an approach to model structure identification illustrated with the Extended Kalman Filter in Beck and Young (1976), and Beck (1979). Two further comments are pertinent. The approach suggested here clearly parallels the proposal made earlier in the response to question A5. Model structure identification also addresses questions similar to the following: should a multiplicative or a threshold growth hypothesis be employed for the description of phytoplankton growth?

The problem described above with respect to Figure 7 is closely related to one interpretation of the notion of self-organization depicted in Figure 8, where in fact we have reduced the self-organization problem to a self-adaptation problem. A six state-variable system is envisaged. At time t_0 the structure of the system is given by the connections indicated in Figure 8a. Some time later, t_i, a recursive estimation of the parameter values, which here are assumed to be time varying, reveals an apparently different structure: a number of the parameters are now estimated as being not significantly nonzero, while other parameters previously zero at t_0 have assumed significantly

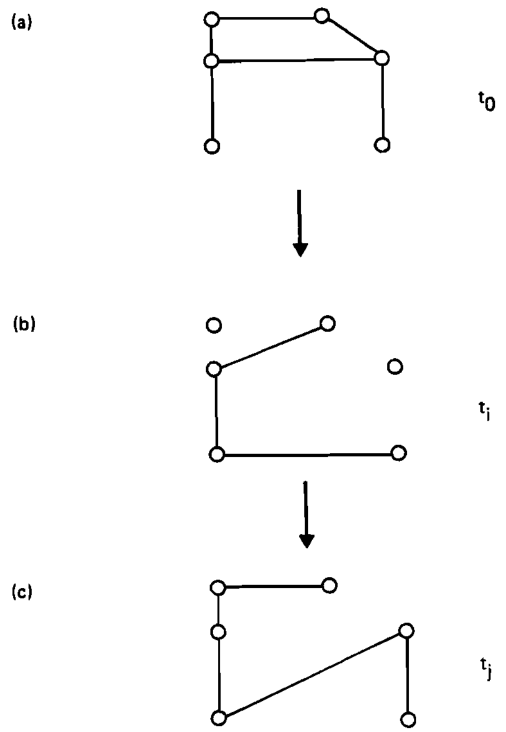
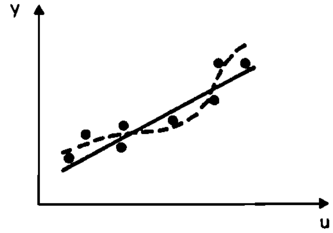


Figure 8. A conceptual picture of self-adaptation (self-organization).

nonzero values (Figure 8b). Further changes can follow at time t_j (Figure 8c), so that apparent structural changes in the ecosystem are accounted for by time-varying changes of the parameter values between essentially negligible and substantially significant values.

Stripped of any sophistication, model structure identification is analogous to the problem of choosing to fit a straight line or a curve (e.g., quadratic, cubic, quartic) to a set of field data, Figure 9a. And while on the subject of simple regression analysis such as this, it might be useful to try to dispel an illusion about parameter estimation. The illusion is that, say, twenty observations allow the determination of twenty pieces of information--i.e., parameter values. In principle there is no

(a)



(b)

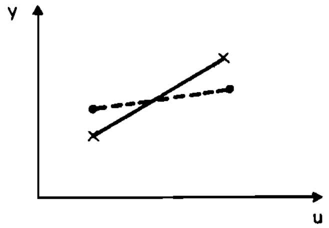


Figure 9. Curve-fitting: (a) the analogous problem of model structure identification and (b) the effect of measurement error.

illusion, but in practice there is. Suppose that the true relationship between u and y in Figure 9b is the continuous straight line drawn through the two observations denoted by crosses. Now suppose that a second set of observations for y is given by the dots, in which the observations contain a small amount of measurement error, and through which an equally feasible straight line could be drawn. Comparing the two sets of measurements it is clear that quite different values would be obtained for the gradient and intercept (the parameters) of the two straight-line fits. The same problem occurs when twenty simultaneous (deterministic) algebraic equations are solved in order to obtain twenty parameter values; and this solution of the problem ignores a primary unwritten rule of parameter estimation: it is necessary to obtain many more observations than there are parameters to be estimated for the analysis to discriminate effectively against the ever-present errors in the measurements.

The final point of this discussion concerned the role of black box modeling approaches; these have received little attention and have perhaps not been thought to be very useful. A black box model identifies merely that which the input disturbances (forcing functions, loadings) can be observed to change in the output responses of an ecosystem. An internally descriptive model expresses the way in which the inputs are related to the state variables, the way in which the state variables are connected among themselves, and the means by which the state variables are translated into output observations. Both types of models tackle the problem of formalizing the relationship between cause and effect. Therefore, at the stage of model structure identification, when the analyst's understanding of observed behavior is incomplete, a black box analysis of cause/effect relationships (e.g., the speed, type, and magnitude of responses to changes) may yield some initial clues about possible corresponding forms of internally descriptive models (Beck 1978). It seems likely that some benefit to the subject of ecological modeling could be gained from a blending of the two approaches.

It might be expected that black box models, with their associated computational simplicity, would be especially useful in a real-time forecasting and control situation. On the other hand, as a response to a question by Markofsky, it was admitted that in general the validity of a black box model does not extend beyond the conditions covered by the sample data set from which the model is derived.

5. *Which lakes or reservoirs should be considered for case studies by IIASA in cooperation with national authorities? Would it be desirable to compare various models having the same data set?*

Reported by J. Fischer

The following comments were made.

Straškraba: There is available a considerable amount of data on the Slapy and Klíčava reservoirs, which has been evaluated by the model CLEANER. Straškraba said that he would be willing to cooperate with any participant who was interested in using the data. Data already used by IBP are also available. In an MIT model used for describing thermal stratification meteorological data relating to the Klićava reservoir are partly taken into account.

Bierman: A comprehensive measurement program has been carried out over several years by U.S. and Canadian authorities to assess the quality of water in Saginaw Bay, Lake Huron. Reports have been published periodically on this case study, and the data is to be transferred to the IIASA data base. (Shortly after the workshop had taken place, this transfer was accomplished.)

Paul: A simple hydrodynamic model was offered for use to the participants. The model works on a steady equations solution basis. It requires only topographic and wind data as input information, and is available at IIASA. This model might also be used for the Lake Balaton Case Study.

Jørgensen: Jørgensen mentioned that his lake model was transferred to the IIASA computer in December 1977.

Finally, it was pointed out that any comparison of models should ideally take place with those people who had developed the models present.

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Part II

PAPERS PRESENTED AT THE CONFERENCE

THE ROLE OF IRON AND CALCIUM IN THE CYCLING OF PHOSPHATES IN SHALLOW LAKES

L. Lijklema and A.H.M. Hieltjes

A close interrelation exists between the physical, chemical, and biological processes in surface waters. Consequently the water quality resulting from these interacting processes generally cannot be understood or predicted from the knowledge of a part of the system, but requires a comprehensive survey of the whole system. However, an overall analysis of the system is often not feasible, and the investigator has to integrate the experience acquired on a set of subsystems studied individually. In this paper such a subsystem, the cycling of phosphates between sediments and overlying water, is discussed in relation to the overall process of eutrophication, in which phosphate is considered to be a key element. The discussion emphasizes the role of inorganic phosphates, in particular of iron and calcium compounds. Figure 1 illustrates some of the main connections between primary productivity and associated decay processes, and the transformation and transport of phosphates.

Inorganic phosphate is converted into particulate organic forms by algal uptake during growth and conveyed further through the food chain. Through decay and hydrolysis a part of the phosphate is regenerated in the water phase, another part will eventually reach the sediments in which also through decay processes--either aerobic or anaerobic--inorganic phosphate is released.

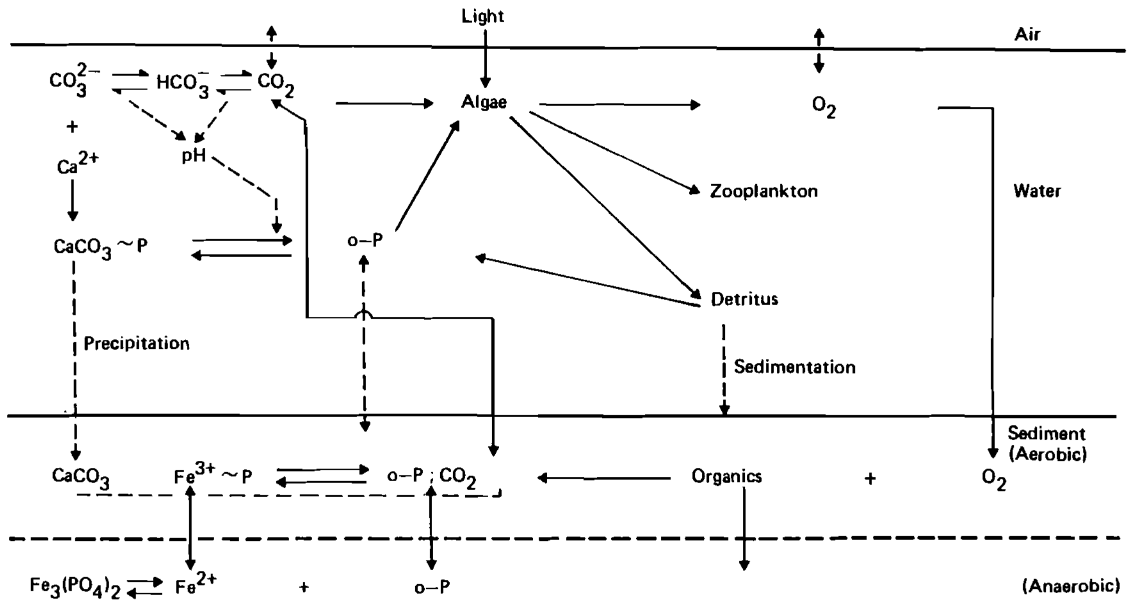


Figure 1. Cycling of phosphates in an ecosystem.

Another process that may result in phosphate transport into the sediments is the precipitation of calcium carbonate induced by photosynthesis. The precipitating carbonate may carry adsorbed phosphate. Although apatite theoretically is the thermodynamic stable mineral in most sediments, in reality iron seems to be the phosphate controlling element in several freshwater sediments. Both pH and redox conditions are of paramount interest for the phosphate adsorption capacity of iron-containing sediments.

PHOSPHATE AND CALCIUM CARBONATE

It has been shown (Stumm and Morgan, 1970) that the surface of calcite can serve for the nucleation of apatite. Figure 2 shows some results of phosphate adsorption experiments on sediments from Lake Brielle, a hypertrophic hard-water lake in the southwest of the Netherlands. These sediments contain about 20 percent calcium carbonate by weight. At high pH considerable

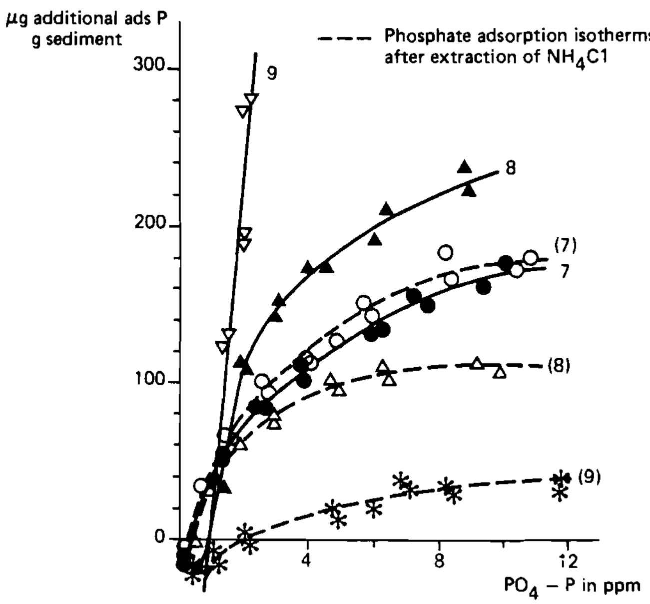


Figure 2. Phosphate adsorption isotherms on Lake Brielle sediments.

amounts of phosphate are adsorbed because of the calcite present; upon removal of the calcite, through ammonium chloride extraction prior to equilibration with a phosphate solution, the adsorption capacity is reduced to much lower values. Also the dependency upon pH is reversed; the highest capacity taking place at low pH, which is characteristic of iron-dominated adsorption. Because biogenic formation of calcium carbonate occurs at comparatively high pH values, the precipitating carbonate may carry substantial amounts of phosphate as a surface complex.

A small laboratory experiment in which the removal of carbon dioxide by photosynthetic processes in lake water was simulated resulted in the formation of a precipitate at pH = ± 9 with concurrent reduction of the orthophosphate concentration from 0.3 to about 0.1 mg/liter. However, upon sedimentation the carbonate is buried in the sediments in which the pH is much lower because of the production of carbon dioxide from the degradation of organic material. In Lake Brielle the pH in the surface water may reach values of 10 and higher, whereas the interstitial water has a pH of around 7 or slightly lower. Hence, a considerable fraction of the carbonates is solubilized

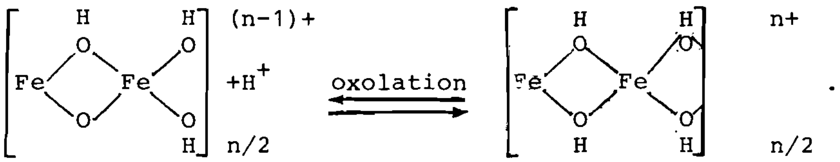
$$CO_2 + CaCO_3 + H_2O \rightarrow Ca^{2+} + 2HCO_3^-,$$

and the bound phosphates are released. The concentration of calcium in the pore-water is generally between 100 and 280 mg calcium/liter, whereas in the overlying water the normal concentration is 80-90 mg/liter. The fraction of carbonate that is not solubilized in the sediments will also lose most of its adsorbed phosphate. Extractions of Lake Brielle sediments containing ammonium chloride solutions result in removal of only 10-15 percent of the total phosphate present. In these extractions the loosely bound phosphates are solubilized, including the phosphate associated with calcium carbonate. Hence, it can be concluded that calcium carbonate may be an important conveyer of phosphate downward into the sediments during periods of high photosynthetic activity, but it plays a minor role in retaining the phosphate in recent sediments of eutrophic freshwater lakes.

IRON/PHOSPHATE INTERACTION

In several anaerobic lake sediments iron (II) has been indicated as the controlling cation in phosphate solubility (e.g., see Emerson, 1976). The evidence is generally obtained by comparing the observed ionic concentrations with the solubility product of vivianite ($Fe_3(PO_4)_2$). Selective extractions of sediment phosphate also often indicate that a major fraction of phosphate is associated with iron. Depending upon the relative abundance of iron and phosphate in the anaerobic sediment, and the presence of other ions and pH, the concentrations of these other ions in the interstitial water are controlled by the solubility of vivianite. By diffusion of iron and/or oxygen and subsequent oxidation, iron (II) may be converted into iron (III), which has a strong tendency to hydrolyze and to form polynuclear species.

$$\begin{array}{ccccccc} Fe^{3+} & \rightarrow & Fe(OH)^{2+} & \rightarrow & Fe(OH)_2^{+} & \leftrightarrows & Fe(OH)_3 \\ & & & & \downarrow & & \\ & & & & Fe_2(OH)_4^{2+} & & \\ & & & & \downarrow & & \end{array}$$



Hydrolysis and polymerization are relatively rapid processes; they are followed by the comparatively slow process of oxolation, in which the hydroxide gradually loses its positive charge (Dousma and Bruyn, 1976). As a consequence the phosphate binding capacity of iron hydroxide is a function of both age and pH. A high positive charge is favored by a low pH and fresh formation, and results in a high binding capacity.

Figure 3 compares adsorption isotherms for the phosphate/aluminum system (pH = 7), which behaves in a similar way to the iron/phosphate system. In one experiment the phosphate was already present when the aluminum was added to the solution (curve 1), whereas in the other the phosphate was adsorbed in a previously formed, but less than 5-minute-old, precipitate (curve 2). The effect of aging is dramatic.

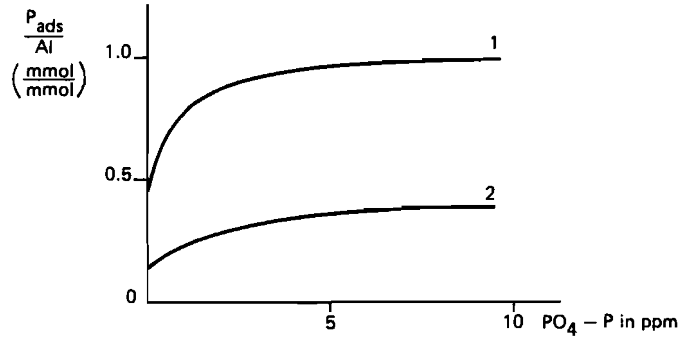
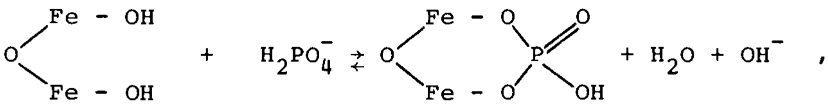


Figure 3. Adsorption of phosphate on aluminum hydroxide (pH = 7). 1 represents *in situ* precipitated aluminum hydroxide, and 2 denotes freshly precipitated aluminum hydroxide.

The effect of pH is shown in Figure 4, representing adsorption isotherms of phosphate on freshly precipitated iron (III) hydroxide. The curves shown fit the datapoints (not shown) well. The curves obey Equation(1):

$$\frac{\text{P adsorbed}}{\text{Fe}} = 0.298 - 0.0316\text{pH} + 0.201\sqrt{[PO_4\text{-}P]} \quad , \qquad (1)$$

in which all concentrations and quantities are in mmol/liter. Underlying this formula is the notion that the first part of the curve, coinciding approximately with the ordinate, represents the high affinity adsorption of phosphate ions on the positively charged sites of the hydroxide. Since hydrogen (H+) is the potential determining ion, this charge will be roughly proportional to log [H+] or pH. The isoelectric point accordingly would be 0.298:0.0316 = 9.4, which is higher than generally accepted. The second part of the curve would correspond to a mass-action-law description for the bidentate sorption of phosphate:



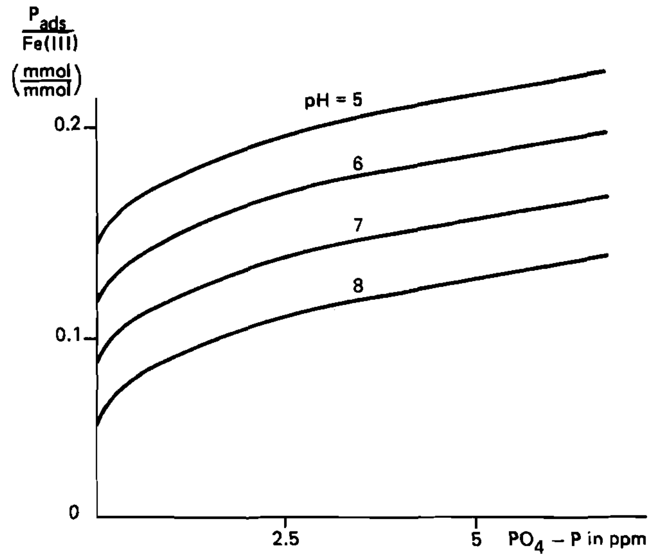


Figure 4. Phosphate adsorption on freshly precipitated iron (III) hydroxide (pH = 5).

resulting in an adsorption proportional to the square root of the phosphate concentration. However, the important fact here is the strong dependence of the phosphate adsorption on pH.

RELEASE OF PHOSPHATE FROM BOTTOM SEDIMENT

The physicochemical information described below is significant for understanding the mechanism of phosphate release from sediments in which iron is predominant. In spring the top layer of the sediment is usually aerobic and contains iron (III) hydroxophosphate complexes in equilibrium with the interstitial water, with comparatively low phosphate concentrations and a pH of 7 or slightly higher. In the anaerobic zone, vivianite controls the interstitial concentration of iron (II) and phosphate; the relative abundance of these elements will be reflected in the ratio of their concentrations. During the summer the anaerobic zone moves gradually towards the water/sediment interface. This is because of increased bacterial activity in the sediments caused by the rising temperature and the influx of readily degradable organic material being deposited as sediment from the overlying water. Also the solubility of oxygen in the water decreases, and temporal stratification may reduce the downward transport.

As long as iron (II) and phosphate diffuse from the anaerobic zone into the aerobic zone of the sediments, the *in situ* oxidation of iron (II), in the presence of phosphate, in an environment with comparatively low pH will result in the formation of iron hydroxy-phosphate precipitates with a relatively high phosphate/ iron ratio. The influx of phosphate into the sediments, as described in the introduction, and the progression of the anaerobic zone towards the sediment/water interface will gradually reduce the favorable conditions for phosphate fixation. The higher overall phosphate/iron ratio and the higher pH in the aerobic zone at the interface will result in the formation of iron (III) hydroxides with insufficient capacity to check the upward phosphate flux. Figure 5 outlines the gradients as discussed above.

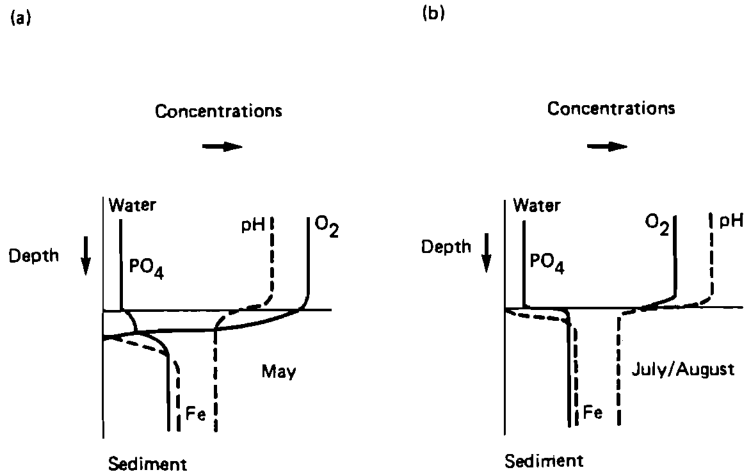


Figure 5. Environmental conditions near the water/sediment interface.

There are field observations supporting this conceptual model. Generally a net phosphate flux from the sediments into the water, as inferred from a phosphate balance of the water, is more or less veiled by inaccurate estimates of the other sources and sinks. However, the data from a potable water reservoir--the Braakman--in the southwest of the Netherlands, which contains sediments similar to those in Lake Brielle, are most significant. This reservoir is filled during the winter; during the remainder of the year there are no other inputs or outputs except for the withdrawal of water and interaction with the (watertight) bottom. Hence changes in total phosphate concentrations are entirely due to exchange with the sediments. Figure 6 shows some observations during recent years. For 1975 and 1976 data on oxygen and pH near the bottom are also available. In 1975 a destratification towards the end of August caused the pH at 0.5 meters above the sediment to rise from 8.5 to 9.5; in 1976 a similar situation occurred towards the end of July. The resulting increase in

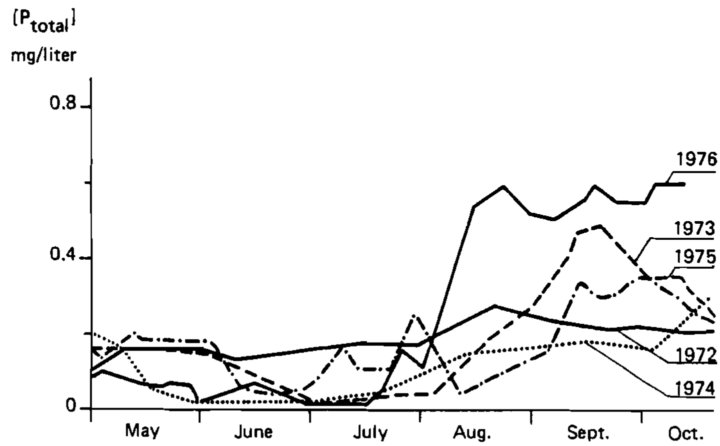


Figure 6. Phosphate concentrations in Braakman II Reservoir.

phosphate concentration can partly be explained by mixing with the hypolimnetic water, but must be due largely to extraction of the top sediment material by high-pH water.

The conclusion is that both pH and redox conditions control the phosphate exchange between sediment and water. Hence, a close interaction exists between primary production and phosphate exchange.

ACKNOWLEDGMENT

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THE PREDICTION OF SOLUTIONS TO MANAGEMENT AND ENGINEERING PROBLEMS RELATED TO THE EUTROPHICATION OF WATER BODIES

V.A. Vavilin, V.B. Vasilyev, M.Yu. Tsitkin, and
S.V. Bagotskii

The eutrophication of water bodies is a typical example of a long-term process that requires predictive estimation. Eutrophication is an increase in the primary production of organic matter, due to the enrichment of a water body with nutrients that stimulate the growth of autotrophic biota. Various natural conditions contribute to the sharp changes that occur in aquatic ecosystems. These conditions are the duration of the warm summer season; the illumination and shallowness of the water--they have a warming effect; little circulation of the water; and the lack of a restraining influence from other factors on algal and biotic production processes.

It is possible to distinguish among the various approaches taken to predict the eutrophication rates of water bodies. The first approach involves finding a correlation between the bloom level and aggregate characteristics, such as total supply of biogenic elements uniting the surface area, average depth of the water body, water circulation rate (Vollenweider, 1968). Another requires the construction of dynamic models (Di Toro et al., 1975; Straškraba, 1975), which may be roughly divided into two groups: small models considering a small number of internal links and external impacts; and large models with a considerable number of variables, internal links, and external impacts.

Large models, whose construction requires time and expense, permit the description of certain structural variations in ecosystems. Small models are useful for predicting solutions to management and problems.

THE DESIGN OF BIOLOGICAL PONDS

An example of the engineering approach is seen in the design of biological ponds, used for the removal of organic matter. An aerated biological pond may be considered as a running water continuous flow culture with ideal water mixing, see Figure 1. The system of equations for concentrations of algae, B, and biogenic elements, N is

$$\begin{cases} dB/dt = -\omega B + (\mu - \beta)B , \\ dN/dt = \omega(N_o - N) - \gamma(\mu - \beta)B , \end{cases} \tag{1}$$

where μ is the specific rate of algal growth; N_o is the concentration of biogenic elements in the influent; ω is the dilution

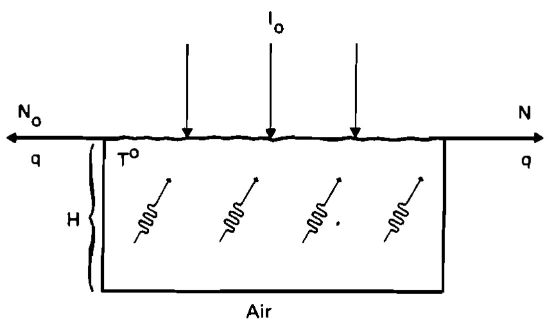

Figure 1. An aerated biological pond. I_o is the surface light intensity; N is biogenic elements; N_o is the concentration of biogenic elements in the influent; q is the influent discharge rate; T^o is the temperature; and H is the full depth of the pond.

rate ($\omega = q/V = 1/T$, where q is the effluent discharge rate, V is the pond volume, T is the liquid retention time in the biological pond); β is the constant representing the rate algal self-oxidation; γ is a stoichiometric coefficient.

Wastewater undergoing additional treatment usually contains a large number of biogenic substances such that their concentration does not affect the algal growth rate. The same is true of dissolved carbon dioxide concentration in an aerated pond. Hence, the main factors limiting the development of algae are temperature, t^o and the light intensity, I. Thus, $\mu = \mu\ (t^o, I)$. The following relationship between μ and the intensity of illumination is commonly assumed:

$$\mu = \begin{cases} \mu_m\ , & \text{if } I \geq I_m \\ \dfrac{\mu_m}{I_m} I\ , & \text{if } I < I_m \end{cases} \tag{2}$$

In order to calculate the average specific growth rate, $\bar{\mu}$, for the whole pond it is necessary to estimate

$$\bar{\mu} = \frac{1}{H} \int_0^H \mu(I(h))dh\ , \tag{3}$$

where h is the depth; H is the full depth of the pond; $I(h) = I_o e^{-nbh}$, where I_o is surface light intensity and n is the coefficient of light absorption by algae.

Let us consider the case of $I_m \leq I_o \leq I_m e^{nbh}$, in which light is a limiting factor at depths h where

$$h \geq h_m = \frac{1}{nb} \ln \frac{I_o}{I}\ . \tag{4}$$

According to Equation (3) we have

$$\bar{\mu} = \frac{\mu_m}{nBH} \left\{ \ln \frac{I_o}{I_m} + 1 - \frac{I_o}{I_m} e^{-nBH} \right\}\ . \tag{5}$$

Under steady-state conditions of further treatment the equation is transformed into

$$\begin{cases} \bar{\mu} - \beta - \omega = 0 \ , \\ \omega(N_o - N) - \gamma\bar{\mu}B + \gamma\beta B = 0 \ . \end{cases} \tag{6}$$

Equation (6) relates the main parameters of a biological pond design (volume, depth, retention time) to the component concentrations in the influent and effluent. Taking into consideration Equation (5), we obtain from Equation (6)

$$N = N_o - \gamma B \ ,$$

$$B = \frac{\mu_m \left\{ \ln\frac{I_o}{I_m} + 1 - \frac{I_o}{I_m} e^{-nBH} \right\}}{nH(\omega + \beta)} \ , \tag{7}$$

where I_o is the average light intensity for 24 hours.

Figure 2 shows estimates obtained using Equation (7) for the following values of constants and parameters: μ_m is αt^o - b; B is ηt^o; α is 0.07 per day/grade; b is 0.28 per day; η is 0.004 per day/grade; γ is 0.1; H is 2 m, 1.5 m, 1 m; I_m is 6,800 lux ; I_o is 216,000 lux (Tashkent, July); t^o is 33 °C; n is 0.0015 mg/liter/cm; N_o is 15 mg/liter.

The calculations show that the effectiveness of further treatment largely depends on the solar radiation, pond depth, and the flow rate.

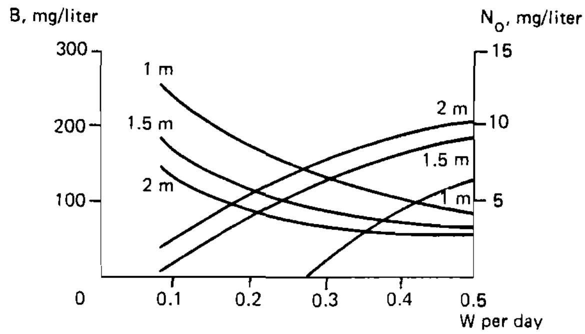

Figure 2. The concentration of the biogenic elements in influent, N_o, and of algae, B, plotted versus the dilution rate, ω.

EUTROPHICATION PROCESSES

Using Equation (1) to describe eutrophication processes in a natural water body, we assume that the concentration of biota is a limiting factor in the process of algal growth. For instance,

$$\mu = \begin{cases} \mu_m \quad , & \text{if } N \geq N_m \quad , \\ \frac{\mu_m}{N_m} N \quad , & \text{if } N < N_m \quad . \end{cases} \tag{8}$$

In case of a natural water body external factors, such as solar radiation, temperature, water circulation rate, supply rate of biotic elements to the water body, have seasonal variations--i.e., they should be considered as variables. However, if the state variables of an ecosystem (phytoplankton concentration and concentration of limiting nutrients) are considered to have rapid responses, the values of state variables will have to be adjusted to the slowly changing values of external factors. In this case, the assumption of steady-state conditions, as represented by Equation (6), is valid.

It is easy to obtain an equation relating phytoplankton concentration to aggregate indices, such as average depth of a water body, supply rate of limiting nutrients, water circulation rate, solar radiation, and temperature--i.e.,

$$B = B(H, \tilde{N}_o, \tilde{W}, \tilde{I}, \tilde{T}^o) \quad , \tag{9}$$

where $\tilde{N}_o$, $\tilde{W}$, $\tilde{I}$, $\tilde{T}^o$ are respective values for a certain season. Since, in the course of the seasonal variation of phytoplankton, spring, summer, and autumn peaks of algal bloom are distinguishable from one another, it is usually necessary to know the values of $\tilde{N}_o$, $\tilde{W}$, $\tilde{I}$, $\tilde{T}^o$ for the bloom peak, as well as the characteristics of the phytoplankton (μ, β, γ).

Versions of Equation (1) may be suggested. If two groups of algae (blue-green, B, and all others, D), two limiting elements (nitrogen, N, and phosphorus, P) and zooplankton, Z, are considered, the equation becomes

$$
\left\{
\begin{aligned}
dB/dt &= -\omega B + (\mu_B - \beta_B)B \quad ,\\
dD/dt &= -\omega D + (\mu_D - \beta_D)D \quad ,\\
dZ/dt &= -\omega Z + (\mu_Z - \beta_Z)Z \quad ,\\
dP/dt &= \omega(P_o - P) - \gamma_{PB}(\mu_B - \beta_B)B\\
&\quad - \gamma_{PD}(\mu_D - \beta_D)D - \gamma_{PZ}(\mu_Z - \beta_Z)Z \quad ,\\
dN/dt &= \omega(N_o - N) - \gamma_{NB}(\mu_B - \beta_B)B\\
&\quad - \gamma_{ND}(\mu_D - \beta_D)D - \gamma_{NZ}(\mu_Z - \beta_Z)Z \quad .
\end{aligned}
\right. \tag{10}
$$

Thus estimation of the biomass of blue-green and all other algae is possible in seasons favorable or unfavorable to blooms.

The construction of complex simulation models is necessary for studies of quantitative changes in aquatic ecosystems, if the concentration of biogenic elements is increasing. In this case, variables must also include various groups of algae (diatomaceous, blue-green, green) and zooplankton (*Infusoria*, *Daphnia*, etc.). Such models allow us to study the possible indirect effect of eutrophication on the various components of an ecosystem.

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TWO EXAMPLES OF THE USE OF HYDRODYNAMIC MODELS IN ECOLOGICAL STUDIES

J.F. Paul

To demonstrate the use of hydrodynamic models in ecological studies, two extreme cases will be described: first, a case in which the solution of the complete three-dimensional hydrodynamic equations was required in order to describe the current pattern adequately; and, second, a case in which no equations had to be solved in order to describe the pertinent processes. These two examples will illustrate that the degree of hydrodynamic detail is dependent upon the particular problem under investigation.

FISH LARVAL TRANSPORT MODEL

The first case involved a study to determine the vulnerability of yellow perch larvae, spawned at different locations in western Lake Erie, to possible entrainment at the water intake from the Detroit Edison Monroe power station (Paul and Patterson, 1977). This particular study formed part of a larger effort to determine the effect of the power plant water withdrawal on the fish population in western Lake Erie (see Figure 1 for a general layout of the fish larval study). The Monroe power plant is the largest fossil fuel generating plant in the United States (3,200 megawatts), and it operates a once-through cooling system. At 100 percent pumping capacity, the plant withdraws 100 m^3/sec of

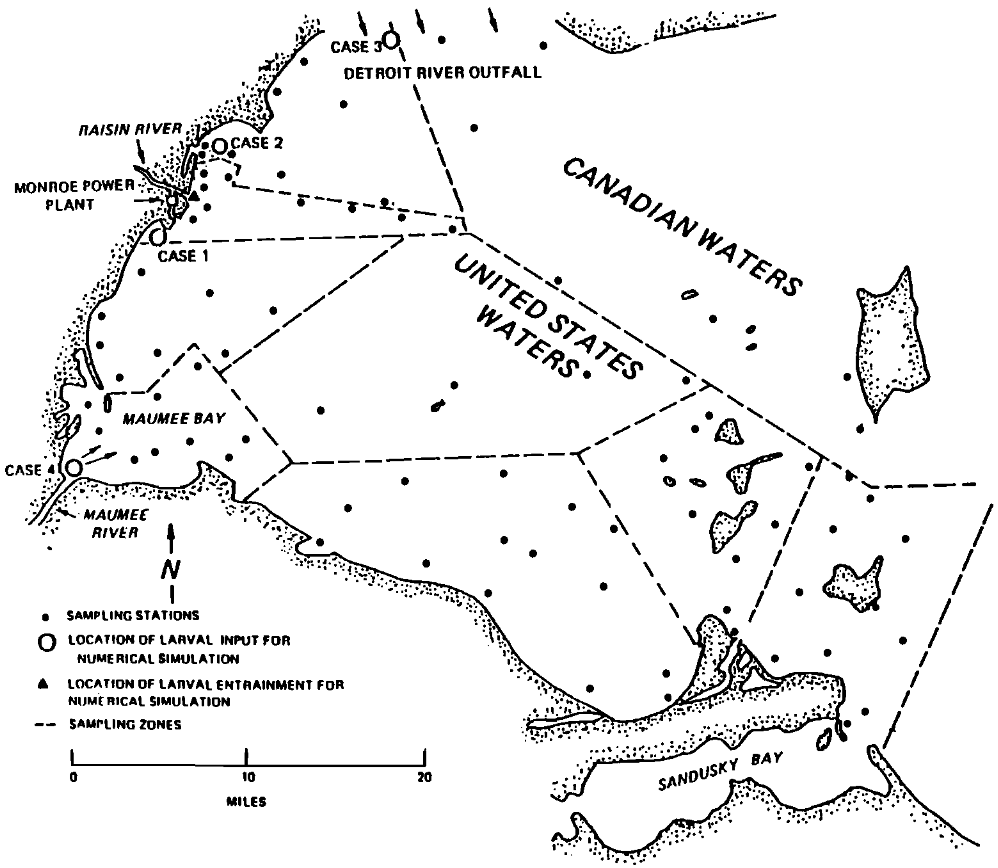


Figure 1. Western Lake Erie fish larval study.

water from the Raisin River. Because the river flow was generally not sufficient to provide for the cooling requirements of the plant, between 5 and 95 percent of the water comes from the lake.

For this study, the steady-state, three-dimensional hydrodynamics were calculated for the typical winds encountered over western Lake Erie in the spring--i.e., south southwest at 6 m/sec. For the initial phase of the study, it was necessary to understand what occurred, under typical conditions, when the fish larvae were spawned and most abundant. Using the calculated three-dimensional currents, the time-dependent transport of the larvae was estimated, assuming that the larvae were spawned at differnt locations in the lake. The three-dimensional, time-dependent, convective-diffusive equation was used to calculate the larval transport. It was assumed that the larvae were transported with the local fluid velocities, except in the vertical direction. It has been observed that the yellow perch larvae have a predelection for the bottom of the water column. To describe this behavior with the model, a settling velocity was added onto the local vertical fluid velocity. The calculations were made for 10 days in 1-hour steps. It has been observed that the larvae acquire exaggerated avoidance mechanisms after 10 days. Since this behavior could not be adequately described by the model, the calculations were stopped at this point. The calculations were made with and without the larval settling velocity, in order to determine the importance of the vertical distribution of the larvae. The percentage of larvae vulnerable to entrainment was calculated for each case.

The calculated surface and near bottom currents are shown in Figure 2. A 1-mile by 1-mile (1.6-kilometer by 1.6-kilometer) horizontal grid with seven vertical points was used in western Lake Erie, and an 8-mile by 8-mile (12.8-kilometer by 12.8-kilometer) horizontal grid was used in the remaining part of the lake. It should be noted that along the Michigan shoreline there are strong northward currents at the surface, but weaker southward currents near the bottom. Also, the surface flow from the Detroit River travels along the Canadian shore, while the bottom flow travels toward U.S. waters.

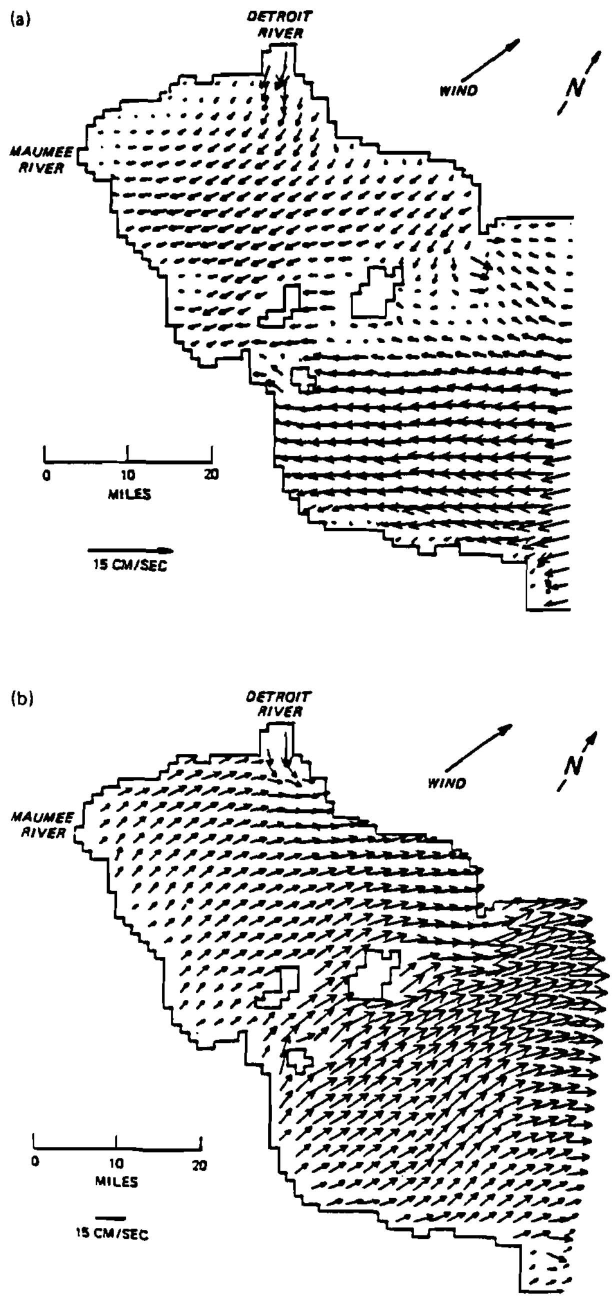


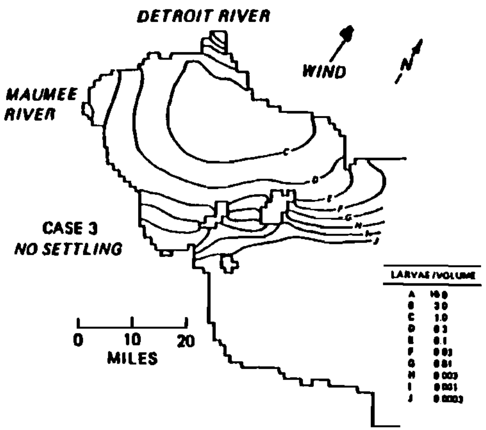
Figure 2. Currents in western Lake Erie for a south southwest wind at 6 m/sec: (a) near bottom currents and (b) surface currents.

The calculated results, using the above-mentioned currents, for the situation in which larvae were initially in the Detroit River is shown in Figure 3. Results for the calculations with and without the larval settling velocity are included. The difference in distribution for the two cases should be noted. The difference between the two cases is particularly evident when the vertically integrated larval concentrations are compared (Figures 3e and 3f). The highest concentration for the vertically integrated larval populations with no settling is confined to the Canadian shore. For the case with settling, the zone of highest concentration moves toward the U.S. shore. These differences in results could not have been obtained if the vertical dependence of the currents had not been included--i.e., if the two-dimensional, vertically integrated currents only were used. This is an example where it was necessary to calculate the three-dimensional currents, in order to provide the flows used in the ecological model.

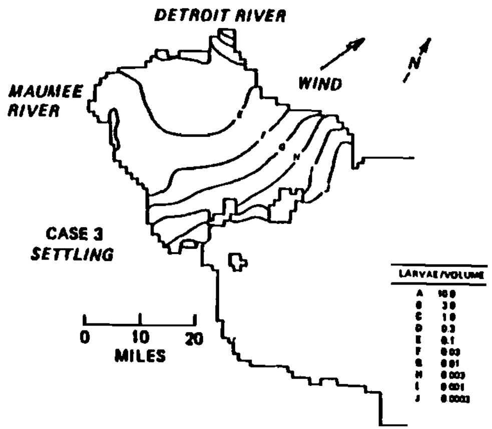
WATER INTAKE PROBLEM

The second case, which did not require a solution of the hydrodynamic equations, was an investigation into the occurrence of taste and odor problems in the water supply at the Saginaw-Midland Water Intake Station at Whitestone Point on Saginaw Bay, Lake Huron (Paul, 1977). During the spring and summer, the intake station usually encounters serious taste and odor problems with the intake water. These problems correspond to the seasons of maximum phytoplankton growth in the inner portion of Saginaw Bay; however, these taste and odor problems do not continue throughout the seasons of the maximum phytoplankton growth. The problem had reached a point where the managers of the intake station were considering a major relocation of the intake pipe, which would be further out into Lake Huron. The present investigation was an attempt to provide an alternative scheme.

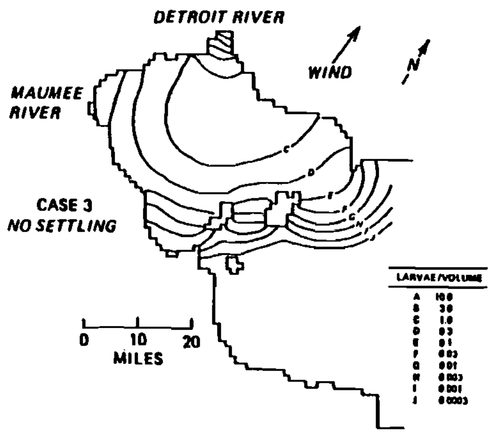
The water intake is located on the northwest shore of Saginaw Bay, 3 kilometers offshore (Figure 4). The inlet is 3 meters off the bottom, in 15 meters of water. The intake is located in the



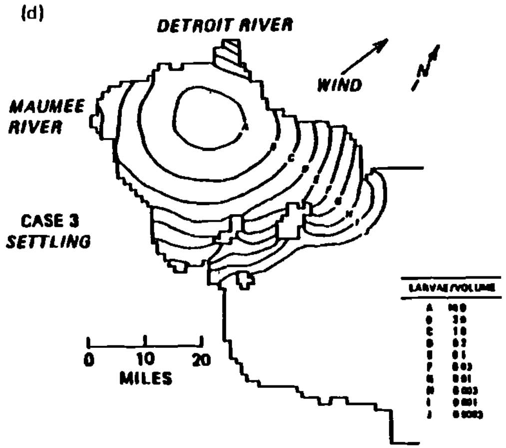
(a) Surface concentration, without larval settling velocity



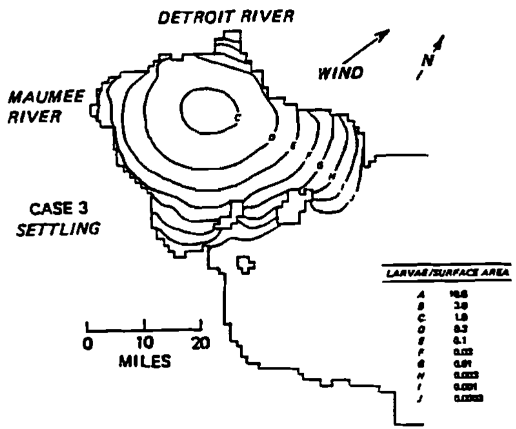
(c) Surface concentration, with larval settling velocity



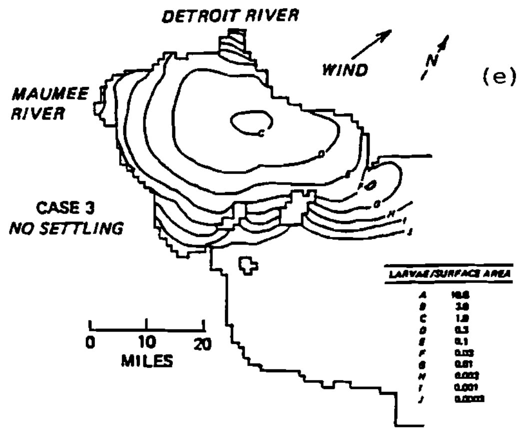
(b) Bottom concentration, without larval settling velocity



(d) Bottom concentration, with larval settling velocity



(e) Vertically integrated concentration, with larval settling velocity



(f) Vertically integrated concentration, without larval settling velocity

Figure 3. Fish larval transport for 10 days after initial implantation in Detroit River.

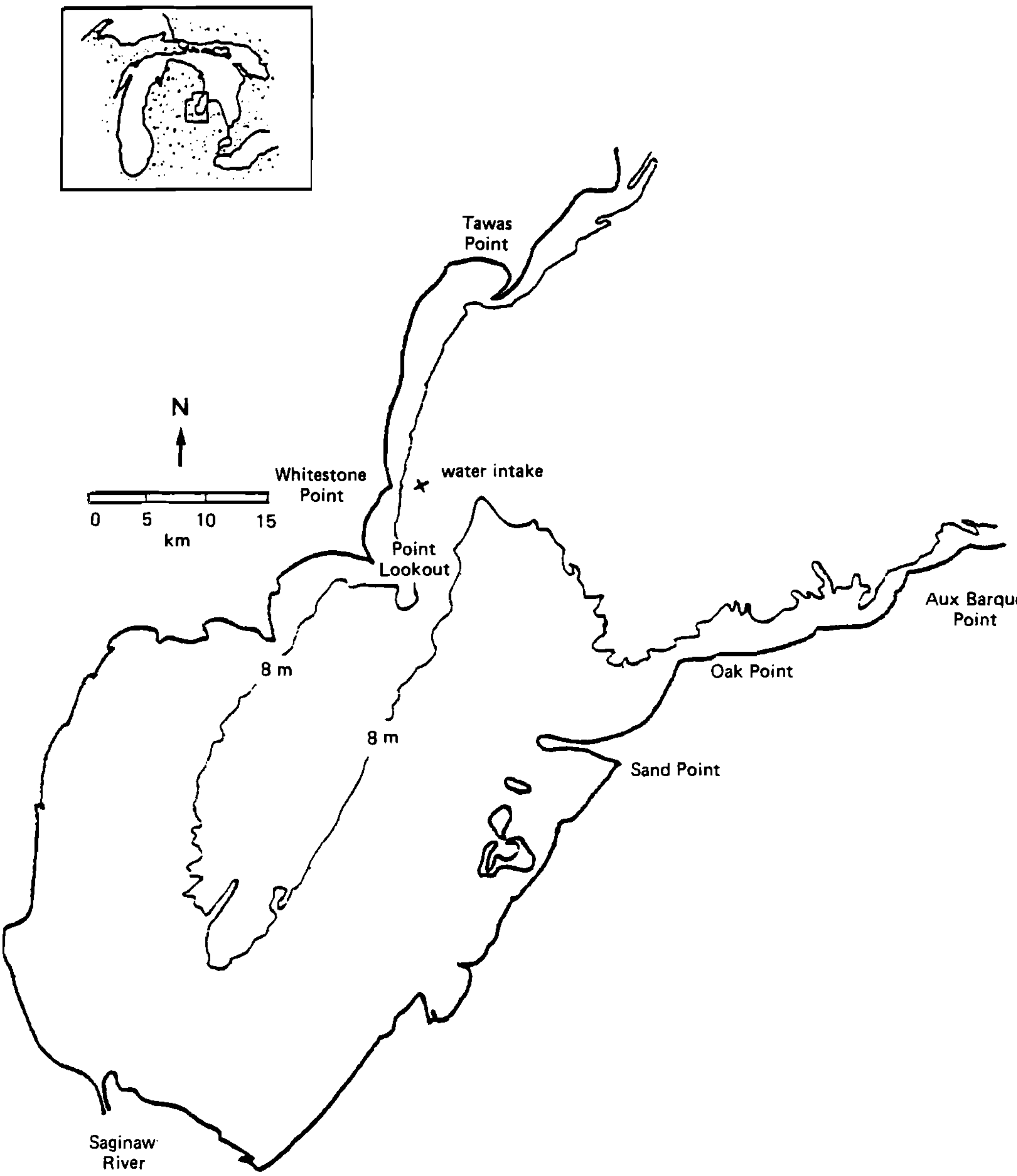


Figure 4. Saginaw Bay, Lake Huron.

relatively deep channel that connects the inner and outer portions of the bay. A field study in the area of intake was undertaken to measure phytoplankton as well as the taste and odor index of the water, and strong correlation between phytoplankton count and odor index was found to exist (Figure 5). The types of phytoplankton dominant in the counts were those known to cause taste and odor problems. From independent field measurements, it was shown that these phytoplankton originated in the inner portion of the bay, the water movement in the bay providing the transport mechanism for these phytoplankton to reach the water intake.

Data on the water intake temperature and wind speed and direction at the Tawas Point weather station were available. A plot of the July 1974 intake water temperature is shown in Figure 6, in which the observed rapid changes in temperature over short periods of time are typical of summer temperature records at the intake. If the corresponding wind directions for July 1974 are examined (Figure 7), it can be seen that the sharp temperature rises correspond to wind shifts to the northwest to northeast range; conversely, the sharp temperature drops correspond to wind shifts from the northwest to northeast range. The circulation pattern is such that when the wind is blowing from the south to west range, water flowing past the intake (which is located near the bottom) comes from the deeper layers of Lake Huron. When the wind blows from the northwest to northeast range, water from inner Saginaw Bay is forced out of the connecting channel and flows past the intake. The general hydrodynamic flows that were deduced from the above information are shown in Figure 8--the flow patterns were confirmed with independent current meter measurements.

There seems to be a correlation between the taste and odor problems occurring at the water intake station and the periods in which the water that is being withdrawn comes from inner Saginaw Bay. A simple solution to the problem is to withdraw the cleaner Lake Huron water. A suggested procedure is to have a variable-depth intake port at the station, and to adjust the withdrawal depending upon the immediate wind direction history.

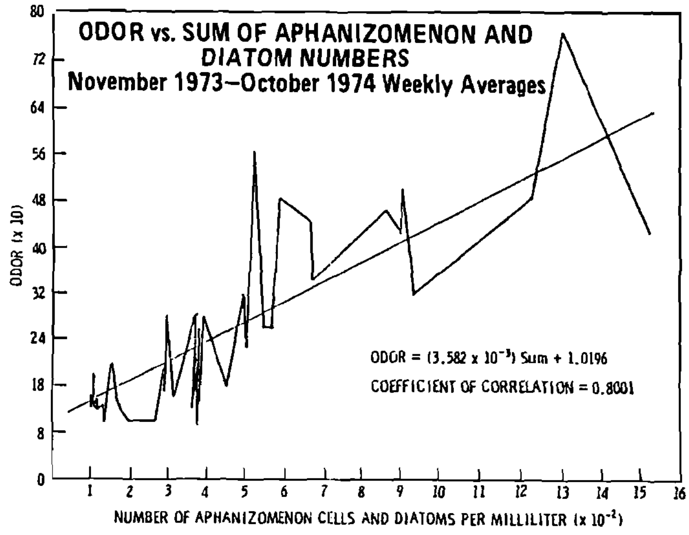


Figure 5. Correlation of phytoplankton count and odor index.

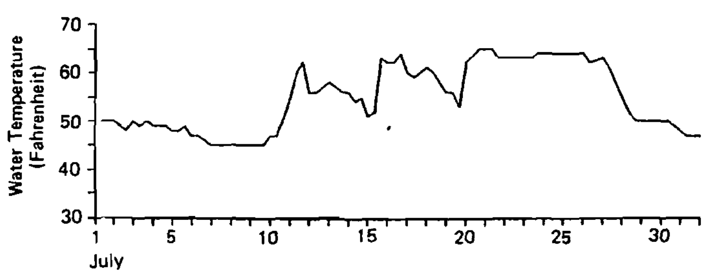


Figure 6. Water intake temperature at East Tawas for July 1974.

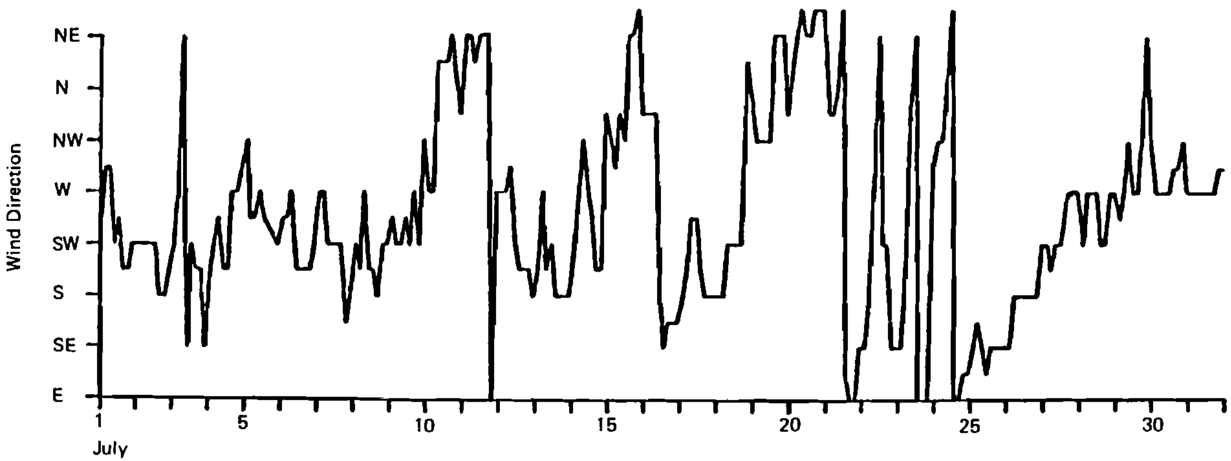
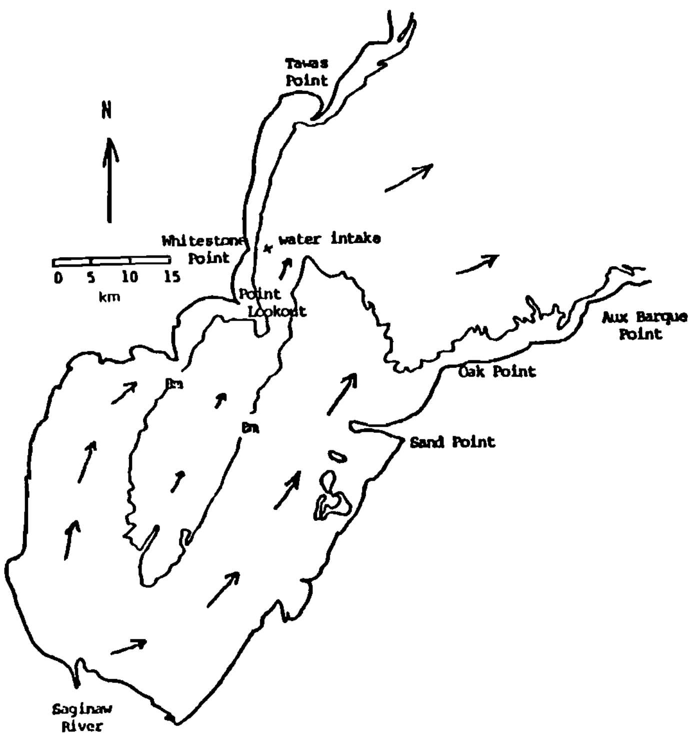
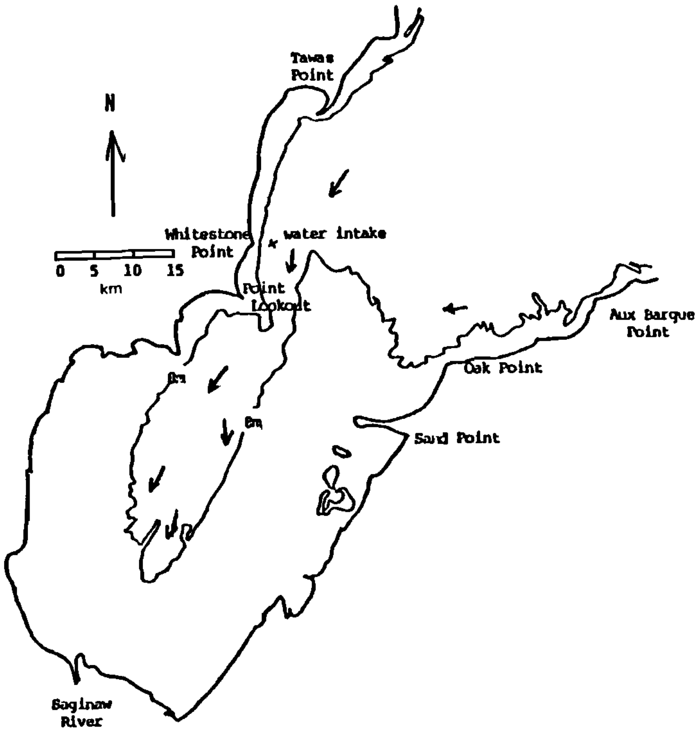


Figure 7. Wind direction at East Tawas for July 1974.

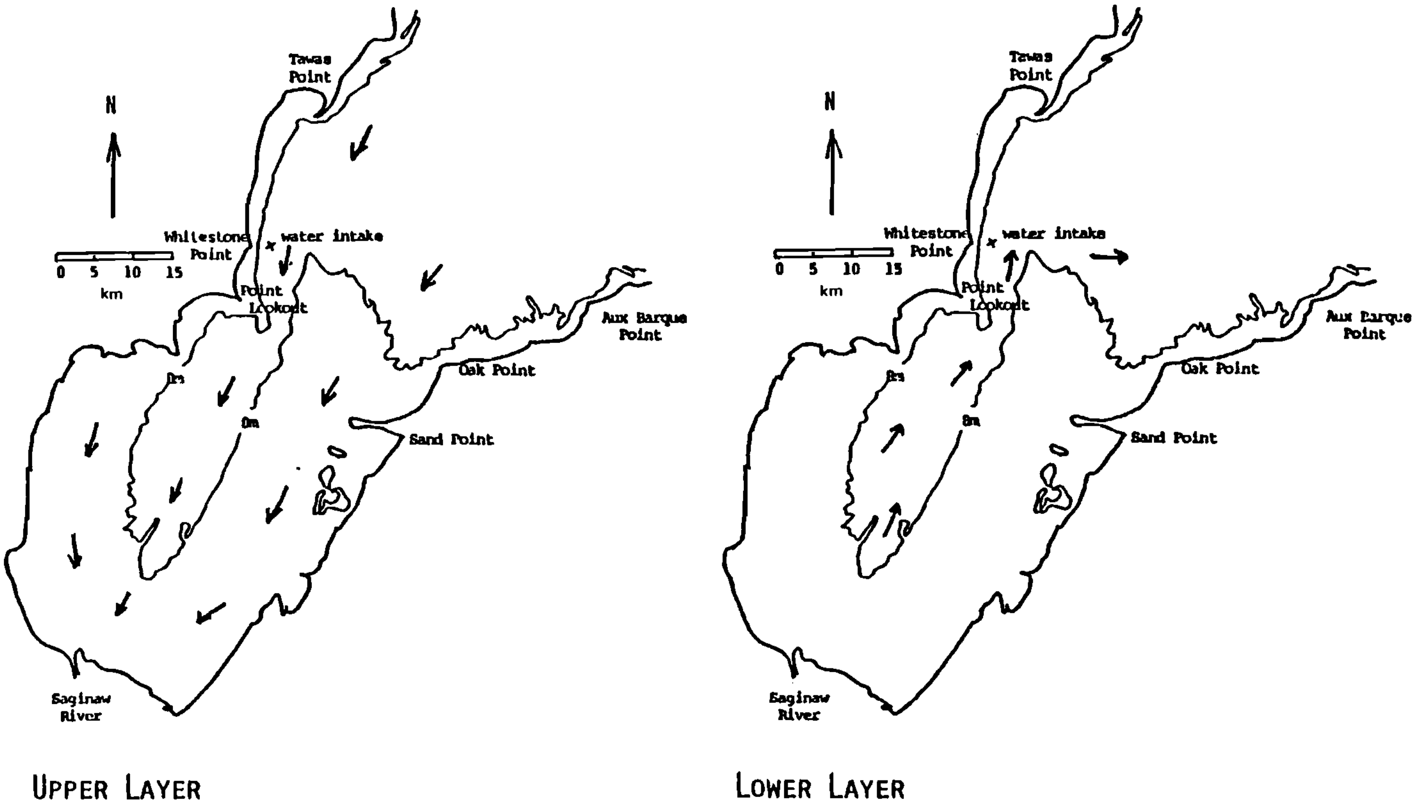


UPPER LAYER



LOWER LAYER

(a) general circulation for south to southwest wind



(b) general circulation for northwest to northeast wind.

Figure 8. Deduced hydrodynamic flows for Saginaw Bay.

No hydrodynamic equations had to be solved. The relevant flow pattern (the hydrodynamic model) was deduced by combining an understanding of basic lake hydrodynamics and available physical data.

SUMMARY

The two examples have been presented to indicate some extremes that may be required in the hydrodynamic details of ecological studies. The detail required in a particular study depends upon the purpose of that study and the related physical mechanisms.

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THE MODELING OF AQUATIC ECOSYSTEMS OF SHALLOW LAKES EXCHANGING MATTER WITH THE SEDIMENT AND WITH NEIGHBORING WATER BODIES

P. Mauersberger

A macroscopic, deterministic model has been developed by Schellenberger, Kozerski, Behrendt and Hoeg in order to study the major features of the aquatic ecosystem of shallow lakes (Schellenberger et al., 1978).

In accordance with the amount of data available for calibration and validation, seventeen chemical and biological components can be taken into account by the cascade model of mixed compartments. The number of the horizontally arranged compartments is arbitrary. Thus, the model is of a "medium size." Other characteristics of the model are its representation of

-- The energy fluxes in the ecosystem
-- Stoichiometric cycling of the nutrients phosphorus and nitrogen
-- Horizontal inflows and outflows of chemical and biological components into/out of the segments
-- Vertical exchange of nutrients and particulate organic matter between sediment and water
-- Solar radiation, water temperature, wind, and external loadings by nutrients and organic matter to act as the "driving forces" of the ecosystem
-- Oxygen saturation of the whole water body, so that the modeling of the oxygen balance does not seem to be necessary

-- Fish and benthic components
-- Time lag between the release of nutrients by organisms and the subsequent re-use by the primary producers
-- Calculation of the hydrodynamic transport processes by a separate simple model

THE CHEMICAL AND BIOLOGICAL COMPONENTS OF THE MODEL

In every segment of the cascade the following constituents are taken into consideration; in the aquatic (pelagic) zone--dissolved inorganic nitrogen, dissolved organic nitrogen, dissolved orthophosphate, dissolved nonorthophosphate phosphorus, detritus (in a special version of the model divided into a labile and a refractory component), two groups of phytoplankton: blue-green algae and "other phytoplankton," zooplankton, fish, heterotrophic bacteria; in the benthic zone--dissolved inorganic nitrogen in the interstitial water, dissolved inorganic phosphorus in the interstitial water, sediment sorbed phosphorus, bottom detritus, phytobenthos, zoobenthos, bottom-populating bacteria. In a special version of the model silica is also taken into account.

The energy content of the biotic and the detritus components is measured and stated in kcal/m^3. The nitrogen and phosphorus concentrations in the organic constituents of the ecosystem (stated in mg nitrogen/m^3 and mg phosphorus/m^3) are treated stoichiometrically. The biomass of a pelagic constituent with an energy content of 1 kilocalorie contains 2.5 milligrams of phosphorus and 18 milligrams of nitrogen. The concentrations of phosphorus and nitrogen in the detritus and the phytobenthos amount to half of these values.

EQUATIONS IN THE MODEL

The model of Schellenberger et al. consists of a system of ordinary differential equations of first order, the solution of which is performed with the help of a digital electronic computer of the type BESM-6.

For the biotic components of the ecosystem (i.e., for the phytoplankton groups, and the consumers and decomposers), the following balance equations are used

$$dB_j/dt = A_j - Q_j - F_j - M_j - Z_j + X_j - Y_j \quad , \qquad (1)$$

where B_j is the energy content of the biomass of the j^{th} component; A_j is the gross production; $A_j = R_j - E_j = \mu_j R_j$ is the gross production of the j^{th} consumer group; E_j is the nonassimilated food; μ_j is the efficiency of assimilation; $R_j = \sum_n R_{jn}$ is ingestion (total feeding rate); R_{nm} is the partial feeding rate (n is prey); $Q_j = c_j A_j$ is dissimilation (respiration); $F_j = \sum_m R_{mj}$ is the grazing rate; M_j is physiological mortality (connected with water temperature and with temperature decline); Z_j is the sinking rate (of plankton); X_j is the external load or/and import from neighboring segments; Y_j is the export into neighboring segments.

The components, B_D, of two groups of particulate organic matter (pelagic and benthic detritus) are increased by mortality, M_j, and by nonassimilated food, E_j, of the aquatic biota, and by the import, X_D, into the segment. They decrease by export, Y_D, by bacterial decay, and through grazing by zooplankton or zoobenthos, R_{kD}. Furthermore, the balance of the pelagic detritus is influenced by the difference, Z_D, between sedimentation and the stirring-up of the sediment, depending upon the temperature, the wind, and the current system. For the benthic detritus, Z_D symbolizes the difference between the losses (from stirring-up or from transition, U, into deeper sediment layers inaccessible to the overlying water) and the sedimentation rate of dead organic matter. Thus, the balance equations of the detritus components are of the type

$$dB_D/dt = \sum_j M_j + \sum_j E_j - \sum_k R_{kD} + X_D - Y_D - Z_D \quad . \qquad (2)$$

The accumulation rate, U, of "stable sediments" is assumed to be

$$U = U_{miner.} + c(B_D' - B_O') \quad , \qquad (3)$$

consisting of the mineral contribution, $U_{miner.}$, and the enrichment of bottom detritus, B_D', exceeding a threshold, B_O'.

In the aquatic zone, the dissolved organic nitrogen and the dissolved nonorthophosphate phosphorus, C_J, are produced proportional to the mortality, M_k, of the aquatic biota, and become reduced by remineralization, $R_j^* = \beta_j C_j A_B$ (A_B is the gross production of bacteria). They are exchanged with the surrounding compartments, X_j, Y_j,

$$dC_j/dt = \sum_k a_{kj} M_k - R_j^* + X_j - Y_j \quad . \tag{4}$$

Dissolved inorganic nitrogen or orthophosphate is generated by bacterial dissimilation, by remineralization of organic nitrogen or "nonorthophosphate phosphorus," and by the nonassimilated food. They are consumed during primary production, $A_m^{\text{prim.}}$, and exchanged with the neighboring compartments

$$\begin{aligned} dC_j^*/dt = {} & \gamma_j A_B + R_j^* + \sum_n \zeta_{nj} R_n \\ & - \sum_m \xi_{mj} A_m^{(\text{prim.})} + Z_j^* + X_j - Y_j \quad , \end{aligned} \tag{5}$$

Z_j^* symbolizes the exchange with the sediments.

SEDIMENT SUBMODELS

A part of the ecosystem model consists of linear balance equations of the following bottom components: dissolved phosphorus and dissolved nitrogen. The variations of dissolved phosphorus in the interstitial water, C_P, result from

-- Bacterial and autolytic decomposition of the bottom detritus and the biotic components that have settled on the sediment
-- The release of sediment sorbed phosphorus, C_P' , enhanced under anaerobic conditions depending on the extent of decomposition of organic materials and the oxygen penetration into the reactive sediment layer
-- Phytobenthic assimilation

-- Chemical fixation and adsorption of phosphorus by the sediment
-- The release of dissolved interstitial phosphorus into the water body

The amount, C_N, of dissolved nitrogen in the interstitial water is also enlarged by the decomposition of bottom detritus and of components dying during sedimentation, but it is diminished by the uptake through phytobenthos, by denitrification, and by release into the free water body. Denitrification is assumed to be proportional to C_N , to the amount of sedimentation of organic material during the previous 30 days, and to a linear function of temperature.

The mixing of dissolved interstitial phosphorus with the sediment (proportional to C_P) is the source of balance in sediment sorbed phosphorus, C_P' , which becomes reduced by resolution (enhanced under anaerobic conditions) and by the transition, U_P', into sediment layers inaccessible to the overlying water body. It is assumed that $U_P' = C_P'U/B_D'$.

A special model for the exchange of phosphorus across the sediment/water interface has been developed by Kozerski (1977), taking into account the following components

-- Organic materials in the reactive sediment layer
-- Iron components in a high level of oxidation
-- Iron components in a lower level of oxidation
-- Phosphates sorbed by oxidized iron components
-- Phosphates sorbed by reduced iron components
-- Chemically fixed phosphorus
-- Dissolved phosphates in the interstitial water
-- Dissolved phosphates in the overlying bottom water
-- Dissolved iron components in the interstitial water
-- Dissolved iron components in the overlying bottom water

and the following processes

-- Sedimentation and decay of organic materials
-- Desorption, reduction and oxidation processes of iron
-- Adsorption of phosphates to iron compounds

- Chemical reactions under varying oxidizing and reducing conditions
- Transport processes between the sediment and the water
- Transport of oxygen into the sediment
- Transition of materials from the thin reactive surface layer (5 to 10 millimeters) into deeper sediments inaccessible to the overlying water body

The adsorption is described by the adsorption isotherm of Langmuir, while all the other processes are simulated by linear relationships (first-order reactions). This model was successfully tested against the laboratory results obtained by Tessenow (1972). But it does not yet allow for rapid environmental variations. Furthermore, accumulation and/or release of sediment nitrogenous substances, resuspension of materials from the sediment by macrobenthos, and the like are beyond the scope of this model.

A fairly general sediment model should comprehend a detailed description of biological, chemical, and physical processes within the sediment and at the sediment/water interface. The description should include

- The influence of rapidly growing or changing benthic populations
- Nonlinear biological and chemical processes and their dependence particularly upon temperature, but also upon redox potential, pH, iron and calcium concentrations, and the like
- The complete system of chemical reactions between the most important nutrient components, including, for example, denitrification
- The sorption and binding capacities of different types of sediments
- The role of "inactive" allochthonous and autochthonous materials, as well as reactivation processes
- The transition of material into sediment layers inaccessible to the overlying water body
- Diffusion processes in the sediment, and diffusion of the soluble substances through the interstitial water

-- Removal mechanisms at the water/mud interface under varying conditions (such as wind, currents, redox potential, diffusion)
-- The influence of turbulence, temperature, biological activities, and the like on the sedimentation process of phytoplankton and detritus
-- The influence of turbulent wind mixing on the sedimentation rates, stirring rates, resuspension of sediments, oxygen penetration, and the related organic decomposition

It is highly desirable that both the aerobic and the anaerobic states are included in the balance equations.

Laboratory experiments with subtracted sediment samples are useful for the investigation of general laws and the evaluation of "general coefficients" (such as adsorption isotherms, decomposition rates as functions of temperature, oxidation and reduction rates, the influence of catalyzing substances).

SPECIAL FEATURES OF THE EQUATIONS IN THE MODEL

Constant Stoichiometric Models

These are suitable for the simulation of the mean annual variations. But, an element-cycle model seems to be necessary for modeling fast processes occurring within a short time interval, for instance during a period of intensive growth. In this case the intercellular concentrations of nutrients should also be taken into account. Obviously, this involves the introduction of many more state variables and parameters into the model, the mathematical structure and practical use of which become more and more complex.

The Uptake of Nutrients by Phytoplankton

This is a nonlinear process described by the Monod kinetics. In the model of Schellenberger et al., the grazing term for each of the consumers is simulated by an exponential relationship

$$R_j = R_j^*[1 - \exp(-a_j \sum_n k_{jn}B_n/R_j^*)] \quad , \tag{6}$$

$$R_j^* = b_jF(T)B_j \quad , \tag{7}$$

or by an equation of Monod's type

$$R_j = R_j^* \sum_n k_{jn}B_n/(\sum_n k_{jn}B_n + R_j^*/a_j) \quad , \tag{8}$$

where R_j is ingestion; B_j is the biomass; T is the temperature; a_j and b_j are species-specific coefficients; and k_{jn} is the trophic relationship matrix. Equation (6) involves the asymptotic cases: $R_j \approx R_j^*$ for optimal feeding conditions, and $R_j \approx a_j \sum_n k_{jn}B_n$ for scarce food concentrations. The half-saturation value in Equation (8) is not assumed to be constant--in accordance with experimental data and to a statement by Holling (1966).

The formulation of the grazing relations is not intended to take care of all specific properties of the different species. Within the framework of an ecosystem model of "medium size," it seems better to use a general formulation of the grazing rate applicable to all types of trophic-level consumers, especially since the loss rates, such as nonpredatory mortality and sedimentation, are also modeled in an approximative manner. But if we aim to model special features of energy and mass transport through the food chain between phytoplankton and fish, we have to take details of the feeding processes into account and must use more specific mathematical descriptions.

The Modeling of Two Groups of Phytoplankton

Schellenberger et al., use the same type of equations, but different parameter values for the two components blue-green algae and others. Furthermore concerning nitrogen fixation by blue-green algae, if the concentration of dissolved inorganic nitrogen falls below a certain threshold, it seems worthwhile, from an ecological and a mathematical point of view, to investigate the splitting of zooplankton into at least two and phytoplankton into more than two components (e.g., a blue-green algae component without nitrogen fixation or diatoms). The modeling of algal successions during the year calls for a "many-component model," but also requires further limnological investigations, in order to obtain data for the calibration and validation of the model. Thus, chemostat data are more useful than batch culture measurements. But since it is difficult to relate experimental findings to natural conditions, careful field investigations are also necessary.

Primary Production by the Phytobenthos

In many eutrophic lakes this is less than the phytoplankton production. Therefore, Schellenberger et al. assume that only the upper nonshaded parts of the submerged plants assimilate nutrients. Thus, the growth of the phytobenthos does not depend upon the biomass of the phytobenthos, but upon the size of the area over which the phytobenthos grows, and upon the optical properties of the overlying water body.

APPLICATION OF THE MODEL

The model was initially developed, calibrated, and verified using data obtained from a chain of shallow inlets along the Baltic Sea coast between Rostock and Stralsund, occupying an area of 197 square kilometers. The mean depth is 1.7 meters. In modeling this chain of inlets, four compartments were used. The input from the drainage basin ($9.7 \cdot 10^5$ m^3/day) and the exchange with the brackish water of the Baltic Sea are taken into account.

Results of the modeling of shallow inlets are discussed in a paper by Schnese and Schellenberger (1976). In each of the model compartments the simulated annual variations of phytoplankton, zooplankton, and nutrient concentrations correspond well to observations (Figure 1). Because of the rapid recycling of phosphorus, the annual variation of the dissolved inorganic nitrogen is much more pronounced than the variation of orthophosphate (Figure 2). Almost 80 percent of the imported phosphorus is shown to be retained in the sediments of the inner inlet regions.

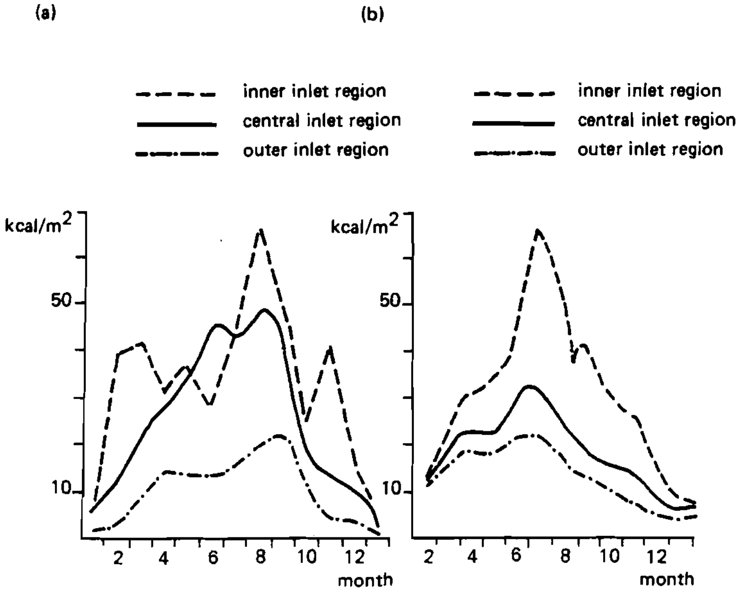

Figure 1. Annual phytoplankton fluctuations: (a) observation and (b) simulation.

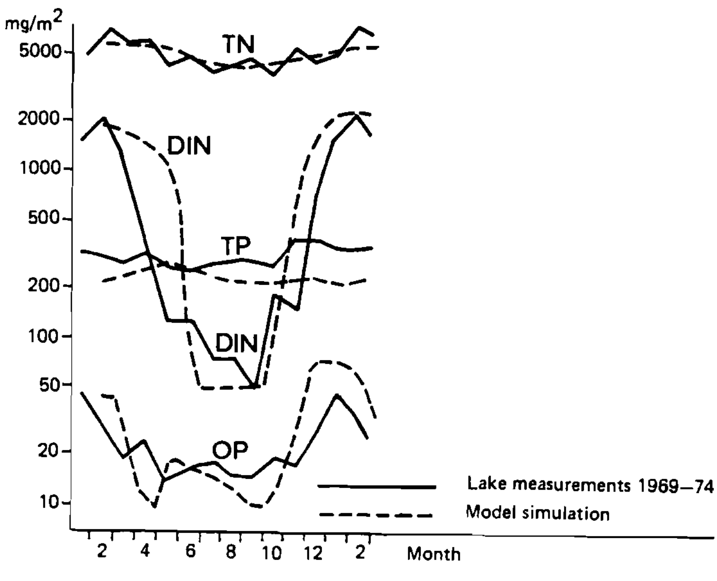

Figure 2. Annual variation of the nutrient concentrations. TN is total nitrogen, TP is total phosphorus, OP is orthophosphate, and DIN is dissolved inorganic nitrogen.

The model has also been applied to the aquatic ecosystem of the Grosser Müggelsee. The River Spree with an average discharge of $8.7 \cdot 10^5$ m^3/day runs through this lake (surface 7.7--square kilometers; mean depth--4.8 meters), which is situated near Berlin, GDR. Some results of modeling the Grosser Müggelsee are shown in Figures 3 and 4, and predictions of the effects of environmental management on the lake's aquatic ecosystem are expected in the future.

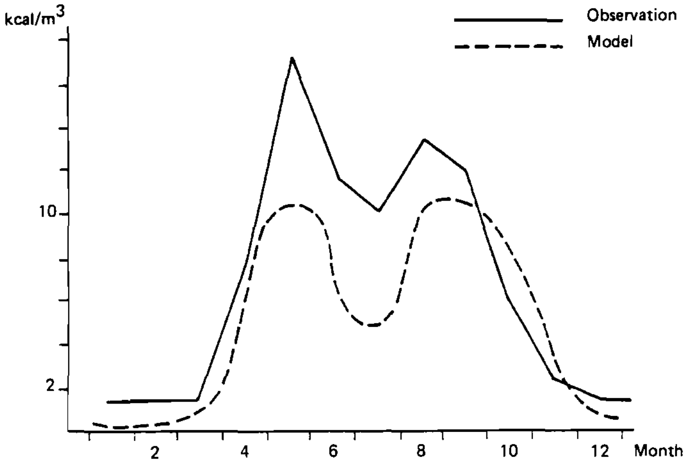

Figure 3. Phytoplankton (blue-green algae and others) in the Grosser Müggelsee.

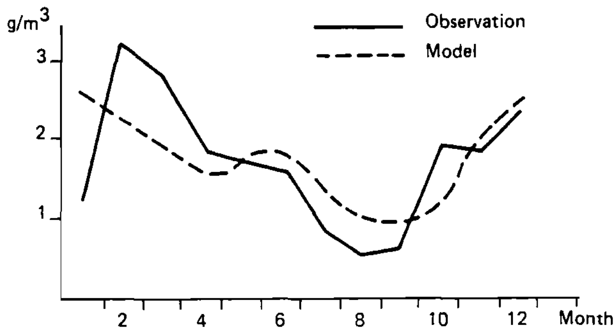

Figure 4. Dissolved inorganic nitrogen in the Grosser Müggelsee.

ACKNOWLEDGMENT

The author is grateful to Drs. Schellenberger and Kozerski for valuable discussions of their model and of general problems of ecosystem modeling.

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DEVELOPMENT AND VERIFICATION OF A MODEL FOR PREDICTING EUTROPHICATION TRENDS IN CZECHOSLOVAKIA

M. Straškraba

An analysis, based on an empirical/theoretical model, of the data gathered during the international synthesis phase of the International Biological Programme--Section Productivity of Freshwater (Straškraba, 1978a) suggests some theoretically sound and empirically verified relationships relevant to eutrophication problems.

A shallow lake is defined as a lake with no permanent temperature stratification. According to an empirical relationship observed for different sets of temperate lakes, the mixing depth is related to the surface dimensions of the lake as characterized by the maximum length, effective length, or, more precisely, fetch size. The empirical relationships imply that a lake with a maximum depth of a few meters will be "deep," if small, whereas a large lake with a depth of 20 meters will be "shallow." The increase in mixing depth as the dimensions of the lake increase causes a hyperbolic decrease in the photosynthetic activity of plankton, and consequently of the chlorophyll concentration.

EUTROPHICATION MODELS

The phytoplankton and chlorophyll-a production concentration of different sets of temperate lakes is related to the total phosphorus or orthophosphate phosphorus concentration by a logistic type of equation, rather than by the Sakamoto, Dillon, and Rigler power curve (Straškraba, 1977, and Desortová et al., 1977). An analysis of fourteen data sets from all over the world (Straškraba, 1978b) suggests that the phosphorus limitation is exchanged for light limitation at total phosphorus values of about 40-70μg/liter. In accordance with the photosynthetic integral concept of "optical depth" as a product of light extinction and mixing depth, shallow and transparent lakes tend to have higher chlorophyll concentrations than deep lakes or less transparent ones.

In the analysis produced by the International Biological Programme, eutrophication is considered to be a multispace process determined by at least the following axes: geographic location, depth, light extinction, limiting nutrient concentration, water retention time, and an exploitation pressure on phytoplankton, mediated particularly through manipulation of fish populations (Straškraba, 1978b).

An analytical simulation model was built to deal with the multidimensionality of the problem. The basic state variables are phytoplankton, zooplankton, and phosphate phosphorus, with feedback interrelationships and nonlinear relationships of photosynthesis and other rates to light, temperature, nutrients, and biomass (Straškraba, 1978b). Three stages of vertical model complexity can be distinguished (Straškraba, 1976b), the one-layer or shallow lake (epilimnion) model, the two-layer or deep lake (epilimnion/hypolimnion) model and the three-layer model (epilimnion/hypolimnion/sediments). Whereas, we have some experience in using the one-layer (Dvořáková, 1974; 1977) and the two-layer model (Dvořáková, 1976), we have no experience in using the three-layer model, which has only recently been developed with the cooperation of Dr. Kozerski, (Institut für Geographie und Geokologie, Akademie der Wissenschaften, GDR), and is at a stage of verification.

We have tested more complex models (Straškraba et al., 1977) but have found their behavior difficult to understand. With the cooperation of Dr. Park's group (Troy, New York, United States), we have modified their complex ecological model, CLEANER, to fit the conditions in Slapy Reservoir, but these modifications are not adequate.

The most important verification procedure of a model is to cover the significant trends of eutrophication that are recognized empirically. Because this involves different processes that must be understood in order to select optimal management schemes, the output of our annual simulation runs includes not only the integral value for the state variables but also the annual changes of the different processes--e.g., gross photosynthesis, respiration, sedimentation, and grazing by zooplankton, rates of phosphorus exchange due to input/output, and to uptake by phytoplankton. Also, the degree of limitation by phytoplankton photosynthesis, due to different variables--e.g., light, temperature, nutrients, the biomass effect--are plotted. The overall change in the state of the lake should correspond not only to observation but also to the different processes taking place in the ecosystem.

This study has recently led us to the hypothesis of a hierarchical switching mechanism for the natural control of eutrophication, which accommodates the following factors:

- Phosphorus loading control of low phosphorus inputs
- Control by depth distribution of light at medium phosphorus loads in deeper lakes
- Control by self-shading and biomass effect at low depths and high phosphorus and hydraulic loads
- Control by zooplankton grazing and internal nutrient cycling (particularly, by zooplankton excretion) at high phosphorus, but low hydraulic, loads
- Control by fish via zooplankton mortality and grazing-rate changes at both average and high loads

In order to compare model runs with the observed trends, cyclic stability of the model must be achieved, and only stabilized runs with self-established initial conditions are evaluated. An automatic program for such plots runs on our Hewlett-Packard 9810 calculator (for a 1-year simulation with the one-layer model, and using a fourth-order constant step Runge-Kutta integration method, the run takes about 30 minutes) and on the IBM 370 (using the simulation language CSMP with a selection of different numerical integration methods, a 1-year simulation takes 1 minute to 1 minute 30 seconds CPU time for the run depending upon the integration method used.

PARAMETER ORGANIZATION

For parameter optimization, the one-layer model is also implemented on a hybrid computer in another computer center with the cooperation of biocybernetic engineers (1-year simulation takes about 16 seconds). Parameter optimization, using both a deterministic optimization procedure for local minima, and, more recently, also a stochastic search of global minima shows that the performance of the model conforms to the observed data. This should not be considered as a guarantee of the accuracy of the model's predictions. Even runs using unsuitable model equations or changed values of nonoptimized parameters are found to fit the data. The hybrid model is also used in two kinds of sensitivity analysis: partial differentiation of the state variables with respect to the parameters, after Tomovic; and a more simple method, involving the representation of the annual trend of differences between standard and disturbed runs normalized to parameter value changes. The latter kind of analysis indicates the restricted value of sensitivity analysis. Mathematically, sensitivity analysis holds for a region of small deviation from the nominal parameter values. This is in agreement with our hypothesis of a switching of internal control during eutrophication; different results from the sensitivity analysis are obtained for the same model and the same parameter values when the forcing functions are varied.

We have concentrated on the analysis of the natural control mechanisms of aquatic systems (Straškraba, 1977), and have formulated a hypothesis of multiple-resource kinetics for the simultaneous limitation of phytoplankton photosynthesis by several factors: light, temperature, nutrient, and biomass (Straškraba, 1978b). This hypothesis suggests that the main drawback of our process equations comes from the use of classical models--e.g., Michaelis-Menten kinetics--which were derived for independent factors, and did not take into account the interaction of these factors. Our attempts to include preliminary quantifications of multiple-resource kinetics into the eutrophication model were not successful, and new models will have to be developed.

In addition to the feedforward and feedback type of control included in most aquatic models, we stress the importance of additional cybernetic control mechanisms: self-adaptation--represented by changes of model parameters according to fixed relations with external control and state variables; and self-organization --represented by changes in model structure. The self-adaptation approach has been investigated (Straškraba, 1976a), but we now feel that this process is only suitable for covering real biological adaptation; for structural changes, other approaches have to be considered. A preliminary attempt to simplify the self-organization problem of the algae in the model into a parameter optimization problem, using methods of dynamic organization, was carried out with the cooperation of colleagues from the Technical University Ilmenau, Section Biocybernetics, GDR (Radtke et al., 1977). The procedure simulates the selection of different species of algae for each instant of time and each dynamic situation by optimizing a time-dependent control variable "individual algal cell volume." The rate parameters of the model are deterministically related to the cell volume by functional relationships. Research to test different optimization methods and goal functions is in progress. A comparison of model runs with direct observation may help to identify the goal functions.

We have recognized the implications of the hierarchical multilayer multigoal control approach by Mesarovic for ecological modeling and hope to formalize the problem in order to solve it on a computer.

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THE MODELING OF NUTRIENT EXCHANGE BETWEEN SEDIMENT AND WATER

G. Jolánkai

In shallow lakes, the nutrient exchange process between the sediment and water is a time-varying phenomenon, or rather, it varies according to meteorological factors, such as wind velocity and wind-induced fluid motion. The bottom sediment is stirred up occasionally by intensive wind and wave action, which causes the sediment particles and the adsorbed nutrients to be entrained, and which allows dissolved interstitial nutrients to enter the water.

SEDIMENT/WATER INTERACTION

Although in calm weather periods there may also be an exchange of nutrients due to diffusion, it may be negligible compared to the scouring effect of agitated fluid motion; this is shown in the extensive fieldwork carried out in Lake Balaton (Tóth et al., 1975) from which the following conclusion can be drawn. While the average rate of phosphorus uptake from the sediment is measured at 0.13 $mg/m^2/day$, it may exceed 30 $mg/m^2/day$ during severe storms. Any value within this range of almost two orders of magnitude may occur. The difficulty in obtaining an accurate value for this model parameter might be overcome in three ways. The most accurate solution is undoubtedly the two-dimensional hydrodynamic modeling of fluid motion and subsequent sediment entrainment, coupled with the modeling of chemical, biochemical, and ecological reactions. This approach, however, involves partly unexplored theoretical problems and parameter determination problems.

The second approach could be purely stochastic, where the only, but significant, limitation is that long records of measurements on nutrient uptake rate are required; these are seldom available.

The third, and probably most suitable, approach involves the application of an aquatic ecosystem (eutrophication) model, whose complexity would depend upon the available data base. The model would use unknown time-varying parameters, including the nutrient uptake rate coefficient. Then calibration of the model, using suitable parameter identification methods, data on the hydrological, chemical, and biological components, and also the measured ranges of coefficient values, would result in a set of calculated coefficient variations for nutrient uptake rate (in the form of a time series). In the next step using this calculated time series for the parameter values (i.e., those values that were "most likely" to have occurred and to have caused the measured changes in the model variables) one may select an appropriate stochastic method to describe the time variations of the selected parameter (e.g., the coefficient for nutrient uptake rate). This approach does not lend itself to purely stochastic considerations, since wind data are generally available, and regression analysis can yield useful empirical formulae on the dynamics of the related processes. Naturally these empirical formulae may also be coupled with stochastic interpretations (i.e., stochastic wind models).

An example of the way in which the above theoretical considerations may be practically applied follows. Assume that fairly long records of algal biomass, dissolved reactive phosphorus, nutrient inputs, and exchangeable sediment phosphorus are available. In a simple algaephosphorus type of eutrophication model (including a parameter for uptake from the sediment), given measured ranges of uptake rate coefficient values, the most probable set of these values is determined by using a maximum likelihood parameter estimation technique. A regression analysis of these estimated coefficient values on past records of other independent variables, such as wind velocity and lake depth, permits an effective prediction of the uptake rate values, providing stochastic models are available for some of the hydrological and meteorological variables.

The above parameter estimation technique and model calibration procedure can be used with other parameters and related independent input variables--e.g., light and temperature limitations on algal growth with the elimination of the limiting equation.

SOURCES OF POLLUTION

Every lake in densely populated regions has four distinct sources of pollution (with the exception of the first source, all are nonpoint sources of pollution)

-- Domestic sewage and industrial wastewater (point sources of sewage treatment plants)
-- Runoff from agricultural and forest land
-- Direct precipitation on the lake surface
-- Urban runoff from cities located along the shoreline

All nonpoint source pollution processes are stochastic in character, and their modeling should be treated accordingly, making use of available hydrological (rainfall runoff) and meteorological models. For more details on the modeling of nonpoint source pollution, see the discussion following question C2.

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SIMULATION MODEL OF AN OLIGOTROPHIC LAKE

J.M. Svirezhev, N.K. Luckyanov, and D.O. Logofet

Oligotrophic lakes are very important as potential sources of high quality freshwater (clean, transparent, and with a high oxygen concentration). But how stable is the water quality in an oligotrophic lake? Because of the low productivity of its ecosystem, the increase in pollution during development of the lake region will not necessarily lead to a corresponding increase in total biomass. One may suppose that the water will continue to become polluted until environmental conditions become unacceptable to the lake's biota, whereupon they will become virtually extinct (the eutrophication state will probably be omitted). This situation will occur under lower pollutant concentrations than in more eutrophic lakes.

Elaboration of a comprehensive mathematical model would permit the forecasting of critical pollution levels and, more generally, the evaluation of the effects of industry on the lake. In this paper we present the preliminary version of a simulation model of an oligotrophic lake's ecosystem. The model was developed from data obtained by observing Lake Onega.

CHARACTERISTICS OF AN OLIGOTROPHIC LAKE

The lake water has a low temperature; this environment is only acceptable to certain phytoplankton species, which are not readily predated by zooplankton. Consequently, another food chain, including bacteria, plays a major role. Decaying phytoplankton are converted to detritus, which is consumed by bacteria, which in turn are eaten by zooplankton.

One feature of an oligotrophic lake is that the inflow of organic matter from land is included in the biological cycle by the bacteria. Another feature is that the oxygen concentration is high at almost all water depths, therefore in simulating the processes taking place in such lakes, it is not necessary to take oxygen concentration into consideration.

MODEL STRUCTURE

The main processes occurring in the ecosystem of an oligotrophic lake can be described by using state variables for phytoplankton (algae), zooplankton, benthos, fish, bacterioplankton (bacteria), detritus, and biogenic ingredients of the environment (nutrients).

The term "concentration" of biological components means the living weight of organisms per unit of water volume. The dependence of growth rate on the temperature is described by the van't Hoff law. It assumes that the ecosystem process could be described by parameters of growth rate; food assimilation coefficients; loss coefficient for the metabolic processes; rates of predation (or of grazing by consumers) described by the Monod formula; and death rates--as functions of temperature.

The diagram of matter flows is shown in Figure 1. It is a "one-point" model (Figure 2), which describes the processes in a certain region of the lake where the parameters could be treated as constants.

Naturally, the ecosystem not only varies in time, but also in space. Thus, the one-point model is not suitable for the simulation of real ecological processes. In the Lake Onega Model,

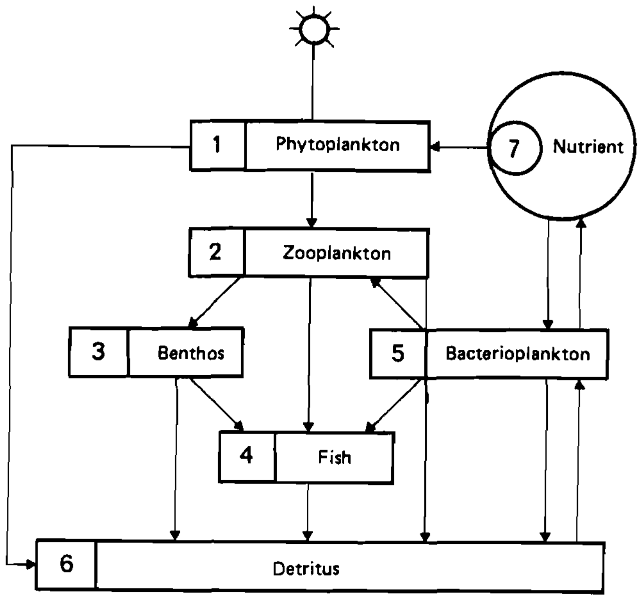

Figure 1. Matter flows in an oligotrophic lake.

we distinguish between five regions with approximately constant hydrological and ecological characteristics. This results in the representation of the entire lake as a system composed of segments.

It was supposed that the concentrations of the components (state variables described by means of differential equations) are the same at all points within a segment. Transition from one compartment to another leads, therefore, to a stepwise change in concentration. The change in concentration of a component due to water flows is equal to the inflow of the component divided by the volume of the segment. The biomass and nutrient dynamics within each segment is described by the same one-point model (but with different parameter values).

The hydrological model applied here is a pure empirical model constructed on the basis of observations of water currents induced by wind.

$$\frac{\dot{x}_1}{x_1} = \mathrm{TPK}\left[\mathrm{RKI}(1)\,\frac{x_7}{10+x_7} - \frac{0.33\,x_2}{H_2(20+x_1)}\right] - 0.33\,\mathrm{DER} \;;$$

$$\frac{\dot{x}_2}{x_2} = \mathrm{TPK}\left[0.33\,\mathrm{RKI}(2)\cdot\left(\frac{x_1}{20+x_1} + \frac{x_2}{1+x_2} + \frac{x_5}{0.3+x_5}\right) - \left(\frac{0.33\,x_2}{H_2(1+x_2)} + \frac{0.25\,x_3}{H_3(0.6+x_2)} + \frac{0.25\,x_4}{H_4(0.03+x_2)}\right)\right] - 0.4\,\mathrm{DER} \;;$$

$$\frac{\dot{x}_3}{x_3} = \mathrm{TPK}\left[0.25\,\mathrm{RKI}(3)\cdot\left(\frac{x_2}{0.6+x_2} + \frac{x_3}{0.03+x_3} + \frac{x_4}{0.24+x_4} + \frac{x_5}{40+x_5}\right) - \left(\frac{0.25\,x_3}{H_3(0.3+x_3)} + \frac{0.25\,x_4}{H_4(0.2+x_4)}\right)\right] - 0.6\,\mathrm{DER} \;;$$

$$\frac{\dot{x}_4}{x_4} = \mathrm{TPK}\left[0.25\,\mathrm{RKI}(4)\cdot\left(\frac{x_2}{0.05+x_2} + \frac{x_3}{0.2+x_3} + \frac{x_4}{0.7+x_4} + \frac{x_5}{10+x_5}\right) - \left(\frac{0.25\,x_3}{H_3(0.2+x_4)} + \frac{0.25\,x_4}{H_4(0.7+x_4)}\right)\right] - 0.5\,\mathrm{DER} \;;$$

$$\frac{\dot{x}_5}{x_5} = \mathrm{TPK}\left[0.8\,\mathrm{RKI}(5)\cdot\left(\frac{x_6}{5+x_6} + \frac{x_7}{200+x_7}\right) - \left(\frac{0.33\,x_2}{H_2(2+x_5)} + \frac{0.25\,x_3}{H_3(40+x_5)} + \frac{0.25\,x_4}{H_4(0.3+x_5)}\right)\right] - 0.5\,\mathrm{DER} \;;$$

$$\dot{x}_6 = \mathrm{TPK}\left\{\frac{x_1\cdot x_7}{10+x_7}\cdot\frac{1-H_1}{H_1} + 0.3\,\mathrm{DER}\,x_1 + 0.33\left(\frac{x_1\cdot x_2}{20+x_1} + \frac{x_2\cdot x_2}{1+x_2} + \frac{x_5\cdot x_2}{0.3+x_5}\right)\cdot\frac{1-H_2}{H_2} + 0.4\,\mathrm{DER}\,x_2 + \right.$$

$$+\,0.25\left(\frac{x_2\cdot x_3}{0.6+x_2} + \frac{x_3\cdot x_3}{0.03+x_3} + \frac{x_4\cdot x_3}{0.24+x_4} + \frac{x_5\cdot x_3}{40+x_5}\right)\cdot\frac{1-H_3}{H_3} + 0.6\,\mathrm{DER}\,x_3 + 0.25\left(\frac{x_3\cdot x_4}{0.05+x_2} + \right.$$

$$\left. + \frac{x_3\cdot x_4}{0.2+x_3} + \frac{x_4\cdot x_4}{0.7+x_4} + \frac{x_5\cdot x_4}{10+x_5}\right)\cdot\frac{1-H_4}{H_4} + 0.5\,\mathrm{DER}\,x_4 - 0.8\,\frac{x_6\cdot x_5}{5+x_6} + 0.5\,\mathrm{DER}\,x_5\Big\}$$

$$\dot{x}_7 = \mathrm{TPK}\left[0.8\left(\frac{x_6\cdot x_5}{5+x_6} + \frac{x_7\cdot x_5}{200+x_7}\right)\cdot\frac{1-H_5}{H_5} - 0.25\,\frac{x_7\cdot x_5}{H_5(200+x_7)} - \frac{x_7\cdot x_1}{H_1(10+x_7)}\right] \;;$$

$$\mathrm{TPK} = 2.^{**}(\mathrm{TP}/10.) \;;\quad \mathrm{DER} = \frac{1+\mathrm{TP}}{0.01+\mathrm{TP}} \;;\quad \mathrm{RKI}(\,) = 1 - \mathrm{RK}(\,) \;;$$

DATA H/0.9,0.6,0.5,0.4,0.4/ ; DATA RK/0.1,0.1,0.2,0.3,0.1/ .

Figure 2. One-point model of the oligotrophic lake ecosystem.

SIMULATION RESULTS

A set of computer runs was made to test the model of an oligotrophic lake having a given limited amount of nutrients. The initial values of the living components were obtained from a study of Lake Onega, and the dynamic behavior of ecosystem components was investigated at a constant temperature. The simulated dynamics of algae, zooplankton and bacterioplankton are shown in Figures 3, 4, and 5. Figure 6 shows the dynamics of the main biological components at a temperature of 15 °C. The results of the runs appear to give a qualitatively correct representation of the main ecological processes occurring in the oligotrophic lake.

AIMS FOR MODEL DEVELOPMENT AND USAGE

The first aim is to refine the hydrobiological data included in the model. A study of Lake Onega is being carried out that includes the collection of data on biological components. This information will give more accurate details of initial conditions. The values of suitable parameters are to be found in the biology literature. If these parameter values are not satisfactory, they will be refined by calibration.

In the future, the effect on Lake Onega of water passing through will be estimated.

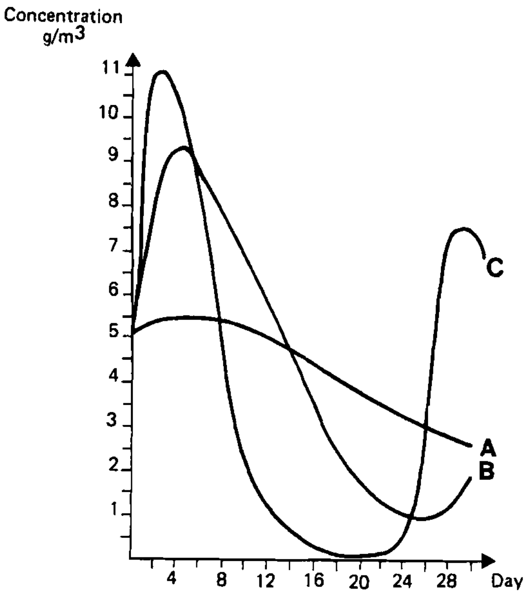

Figure 3. The dynamics of algal concentration. A is 5 °C, B is 10 °C, and C is 15 °C.

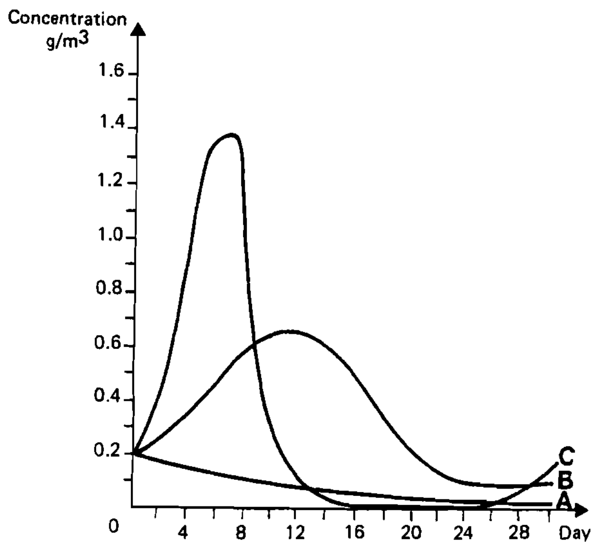

Figure 4. The dynamics of zooplankton concentration. A is 5 °C, B is 10 °C, and C is 15 °C.

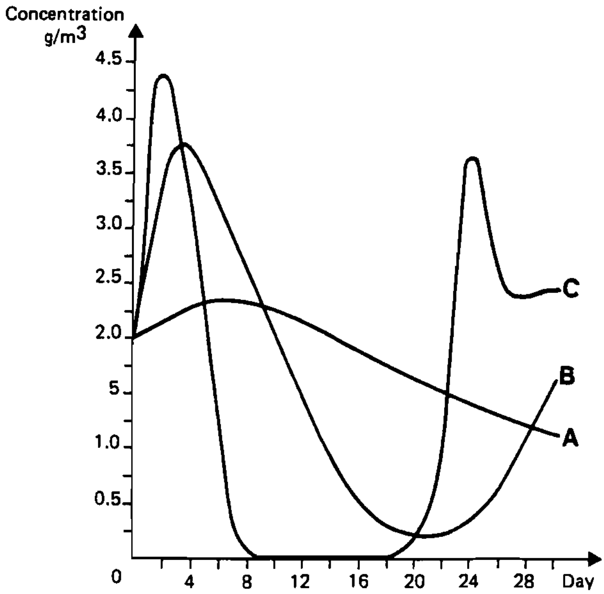

Figure 5. The dynamics of bacteria concentration. A is 5 °C, B is 10 °C, and C is 15 °C.

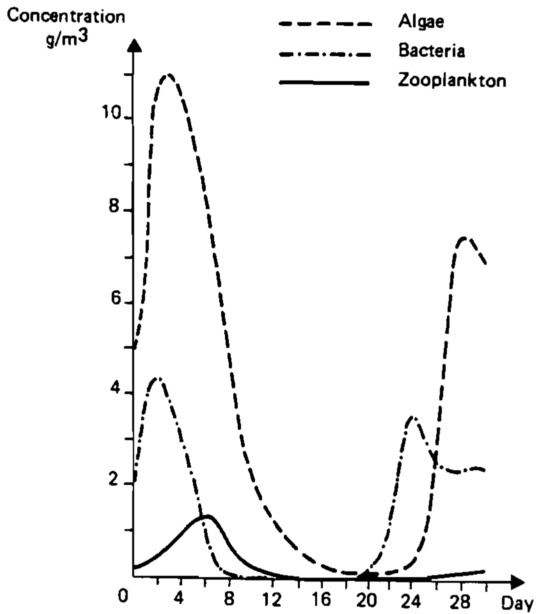

Figure 6. The dynamics of the principal biological ingredients, when the temperature is 15 °C.

EVALUATION OF AVAILABLE PHOSPHORUS MODELS

M.W. Lorenzen and F.M. Haydock

Phosphorus is potentially detrimental to surface waters, because of its role in the eutrophication process. Phosphorus sources include direct rainfall, groundwater inflow, inflow from streams and rivers, direct runoff, domestic waste discharges, and industrial waste discharges.

This preliminary report is intended to provide a review and evaluation of the available models that could be used to analyze the effectiveness of possible options for controlling phosphorus loading. Section 2 reviews methods to compute phosphorus loading rates. Methods evaluated include land use loading rates suggested by Porcella et al. (1974), and Uttormark et al. (1974), and the data base created as part of the National Eutrophication Survey (Omernik, 1977). Section 3 evaluates available lake response models for phosphorus. Three groups of models are considered: one box/one parameter, two box/one parameter, and two box/two parameter. Procedures to relate phosphorus concentrations to maximum algal biomass and water transparency have been reviewed by Lorenzen (1977) and are not included here. Section 4 provides a summary of our preliminary recommendations for analysis of the data base and application of models.

PHOSPHORUS LOADING

Of the three methods reviewed here, only the first takes the effect of point source pollutants into consideration. It is based on general mass flow relationships of pollutants from various natural and cultural (man-related) sources. The two alternative methods are concentrated on nonpoint source nutrient loadings. The procedure used in both cases was to compile and analyze previous survey data to arrive at expected nutrient loadings based on various land use practices.

Porcella et al.

Porcella, et al. (1974) developed a general phosphorus mass flow model to account for pollutant loadings from cultural activities and natural sources. These sources were divided into diffuse and point sources and were classified in one of the following seven categories.

- -- Nonbasin activities (rainfall, river inflow from another basin)
- -- Agricultural (fertilizer and pesticide use, return flows)
- -- Urban and rural watersheds (solid-waste disposal, managed forests, grazed watersheds, undisturbed and undeveloped natural watersheds, urban runoff)
- -- Domestic wastes (human wastes, detergent phosphorus)
- -- Industrial wastes (such as industrial detergents, water softeners)
- -- Mining activities (phosphorus mining and runoff from strip mining)
- -- Animal production (animal wastes from cattle, poultry, pigs, sheep)

The program subroutines above were further classified into four categories based on the relative importance and general nature of the source. The categories consisted of minor diffuse sources, controllable diffuse sources, minor point sources, and controllable point sources. The following sections contain descriptions and definitions of the components of the program.

Minor Diffuse Sources

The phosphorus contribution from the use of organophosphate pesticides is computed as

$$P_{op} = A_{op}F \quad , \tag{1}$$

where subscript op represents the use of organophosphate pesticides; P is the phosphate loading, g/yr; A is the area of land; F_{op} is the estimated annual consumption of organophosphate pesticide in g equivalents of phosphorus acre.

The phosphorus contribution from nonphosphorus mining activities is computed as a function of total strip-mined or mine-tailing acreage

$$P_{sm} = A_{sm}C_{sm}F_{sm}R \quad , \tag{2}$$

where sm represents the strip-mining activities; R is the annual precipitation rate; C is the concentration of phosphorus in the runoff water from this area; F_{sm} is the ratio of the runoff water to the total precipitation. The remaining variables are as defined previously.

The phosphorus output from solid-waste disposal is related to the mean annual precipitation

$$P_{sw} = A_{sw}C_{sw}F_{sw}R \quad , \tag{3}$$

where sw is the solid-waste disposal and land fill activities, and F_{sw} is a runoff factor providing for a maximum rate of runoff when the surface becomes saturated. The remaining variables are as defined previously.

The contribution of phosphorus from direct rainfall onto surface water in the basin may be considered to be significant under certain local conditions. It is computed as

$$P_r = A_rC_rR \quad , \tag{4}$$

where r is the direct rainfall, and A is the total surface area of water bodies in the basin.

Controllable Diffuse Sources

Phosphorus loading as a result of urban runoff is estimated according to the following relationship

$$P_u = A_u C_u F_u R \quad , \tag{5}$$

where u represents urban land use, and F_u is a factor relating runoff to precipitation.

Phosphorus runoff from natual watersheds, developed watersheds, managed forest, and grazing land is computed first by determining a factor related to the saturation of the watershed soil and its runoff rate.

$$F = C_{nw} K_w R \quad , \tag{6}$$

where F is the runoff rate per unit area; C_{nw} is the phosphorus concentration in runoff from natural watersheds; and K_w is a first-order/zero-order type runoff relationship to provide maximum rate of runoff when the surface of the watershed is saturated.

Assuming that developed watersheds produce approximately twice as much phosphorus runoff as natural watersheds, and that managed forests and grazing lands produce approximately one and one-half as much phosphorus runoff, the total estimate of watershed runoff is computed as

$$P_w = FA_{nw} + 2FA_{dw} + 1.5F(A_f + A_g) \quad , \tag{7}$$

where the subscripts are as follows: w is watershed, n is natural, d is developed, f is managed forest, and g is grazing lands. The following equation is used to determine an overall erosion rate for each type of land.

$$Z = E \cdot K \cdot C \cdot P \cdot L \cdot S \quad , \tag{8}$$

where E represents erosivity; K is a factor related to the variation in the soils and properties of erosion; C is the cropping management factor; P is the practice factor, which accounts for contouring, terracing, strip cropping, or other practices reducing erosive potential of runoff; L is the mean length of slope; and S is $0.0076 + 0.0053\ s + 0.00075\ s^2$, in which s is the percentage slope for a particular cropland system.

Three grades of land slope are defined by Porcella et al.: 0-1 percent, 1-5 percent, and 5-10 percent. An average slope within each grade, 0.5, 3.0, and 7.5 percent, is used in the equation. The proportion of land falling within each grade of slope is then estimated, and the erosion rate is computed for each grade. The overall erosion rate is then obtained by summation.

The various forms of phosphorus contributions are then calculated as

$$P_{org} = Z_T F_1 F_2 \quad , \tag{9}$$

where P_{org} is the phosphate derived from organic material in the soil; Z_T is the total erosion; F_1 is the fractional percentage of soil that is assumed to be organic; and F_2 is the fractional percentage of phosphorus in the organic material.

$$P_{inorg} = Z_T(1-F_1)(F_3) \quad , \tag{10}$$

where P_{inorg} is the phosphate derived from inorganic material in the soil, and F_3 is the fractional percentage of phosphorus in the inorganic material per unit area of 15-centimeter depth.

The total phosphorus output from fertilized soil is then computed as

$$P_a = (P_{org} + P_{inorg} + P_{sol})A_a \quad , \tag{11}$$

where a represents agriculture, and P_{sol} is the soluble phosphorus that is considered to be a constant value per unit area as long as sufficient solid phase phosphorus is present.

Minor Point Sources

Irrigation return flow phosphorus contribution is computed as

$$P_{rf} = Q_{rf} C_{rf} \quad , \tag{12}$$

where subscript rf is the irrigation return flow, and Q is the flow.

River flow contributions from outside the drainage basin are considered uncontrollable. Therefore, they are included in this subsection, although, in some cases, they may be significant pollution sources. The mass input is computed by the same formula as given above for irrigation return flow.

Controllable Point Sources

The input of phosphorus from human waste products is calculated on a per capita basis. This total is then subdivided into three disposal categories:

- Septic tank--this quantity is considered as a "sink"
- Sewered, direct discharge--this quantity is considered as a direct source
- Sewered, treatment plant--this quantity is added to the total input of the model of the treatment plant process.

The computation of phosphorus input from both domestic and industrial use of detergents is based on the estimated consumption of four levels of washing products used in these sectors: high phosphate detergents, low phosphate detergents, phosphateless detergents, and soaps.

Mean estimated phosphorus concentrations and mean relative per capita domestic use are calculated for each detergent. The total per capita contribution of detergent phosphorus is calculated by summation, and then multiplied by the population in the drainage basin.

For the model representing phosphorus removal processes, input consists of the total discharge of phosphorus assumed to enter the treatment plant. A part of this model determines the final concentration of phosphorus discharged and indicates the least costly treatment system providing an adequate level of phosphorus removal.

Uttormark et al.

In an effort to provide predictive procedures for estimating nonpoint source nutrient loadings to receiving waters, Uttormark et al. (1974) compiled data from the literature and converted it to a common system of units (kg/ha/yr). In order to develop systems for estimating nutrient runoff, the resulting pool of data was subdivided into four land use categories: agricultural, urban, forest, and wetlands. Data describing nutrient transport were also reviewed, in order to estimate the relative importance of various transport vectors including groundwater, surface water, precipitation, and dry fallout.

By averaging and comparing available data, a set of ranges were computed providing relative comparisons of what might be considered to be high, low, or average values relative to the other numbers in the data set. A summary of typical values of phosphorus runoff coefficients, as reported by Uttormark et al. (1974) is presented in Table 1. These values were obtained by averaging and comparing available coefficients disregarding the location. Therefore, these numbers do not necessarily apply directly to all parts of the United States.

The results of this study showed that there is little justification for the delineation of land usage within direct drainage basins beyond the four categories mentioned above. Nutrient concentrations were found to be highest in urban runoff, and lowest in runoff from forested areas. Various transport mechanisms for nutrient export from agricultural lands were investigated, including seepage through vertical soil profile, overland runoff, and transport by streams draining agricultural watersheds. Transport by streams was found to be the most important mechanism for estimating nutrient loadings to lakes from agricultural sources. Marshes and wetlands were found to act as holding ponds, storing phosphorus during the growing season, and releasing it later; net nutrient runoff from these sources was estimated to be zero. Nutrient contributions from other diffuse sources, including groundwater seepage, rain, and fallout, were found to be large in some areas. Reliable estimates of the loading from these sources can be based only on site-specific information.

Table 1. Typical values of phosphorus runoff coefficients.

	Diss. inorg. phosp. kg/ha/yr			Total phosp. kg/ha/yr		
Type of land	High	Low	Average	High	Low	Average
Urban	2.0	0.5	1.0	5.0	1.0	1.5
Forest	0.1	0.01	0.05	0.8	0.05	0.2
Agricultural	0.5	0.05	0.1	1.0	0.1	0.3

SOURCE: Uttormark et al. (1974)

The information summarized by Uttormark et al. provides an estimate of the relative importance of potentially significant nutrient sources. Further resolution is required in order to compare management alternatives on a site-specific basis.

Omernik

Omernik (1977) developed a method for predicting the nonpoint source nutrient levels in streams. The study was based on data collected as part of the National Eutrophication Survey from a nationwide network of 928 nonpoint source watersheds. These data were analyzed in an effort to correlate drainage area characteristics with nutrient levels in streams.

Omernik reported good correlations, r^2, between general land use and nutrient concentrations in streams. He noted that mean phosphorus concentrations were, on average, nine times greater in streams draining agricultural watersheds than in streams draining forested watersheds.

Two methods were developed for predicting nonpoint source nutrient concentrations in cases where adequate stream nutrient data are not available to a planner, and where a field sampling program is not feasible, because of a lack of time and/or money.

LAKE RESPONSE MODELS

A number of models have been developed to evaluate the effects of phosphorus loading rates on lake water quality. This section reviews the major models that may be relevant to a national assessment of the impact of phosphorus control measures. For each model, descriptions are given of the formulations, input data requirements, output, and prior use. A summary comparison of basic processes considered and their applicability is given at the end of the section.

Vollenweider

Vollenweider (1968, 1969, 1975) has introduced the idea of relating a lake's trophic state to its phosphorus loading rates. Early work developed phosphorus loading curves, but later it was recognized that the effects of hydraulic residence time and the concentration of phosphorus inputs were not adequately accounted for by this method. The effects of hydraulic residence time were taken into account by plotting real phosphorus loading, $g/m^2/yr$, as a function of real hydraulic loading, q_s, m/yr. Vollenweider's "permissable" influent concentration can be defined as

$$C_{ip} = 0.1(q_s)^{-\frac{1}{2}} \quad , \qquad (13)$$

where C_{ip} is the permissable influent phosphorus concentration, g/m^3, and q_s is the surface hydraulic loading rate, m/yr.

A mathematical basis for these relationships can be developed from simple mass balance concepts: for a single-compartment, completely mixed lake,

$$\frac{dC}{dt} = \frac{C_{in}Q}{V} - \frac{SCA}{V} - \frac{QC}{V} \quad , \qquad (14)$$

where C_{in} is the average annual influent phosphorus concentration, g/m^3; C is the average annual phosphorus concentration, g/m^3; V is the lake volume, m^3; S is the sedimentation coefficient, m/yr; A is the surface area, m^2; Q is the annual hydraulic inflow/

outflow, m^3/yr; and t is the time, yrs. Then, at steady state $\left(\frac{dC}{dt} = 0\right)$,

$$C_{ss} = C_{in} \frac{1}{1 + \frac{SA}{Q}} \quad , \tag{15}$$

where C_{ss} is the average annual steady-state phosphorus concentration.

If it is assumed that a "permissible" loading is one that results in a specified phosphorus level (i.e., 20 μg/liter), then a line representing the permissible steady-state concentration for various combinations of phosphorus and hydraulic loading can be plotted. Furthermore, Vollenweider has reported a statistical relationship showing that

$$C_{ss} = C_{in} \frac{1}{1 + \sqrt{\frac{\bar{z}A}{Q}}} \quad , \tag{16}$$

where $\bar{z}$ is the mean depth. Using the same model (mass balance), this relationship introduces an additional parameter, $\bar{z}$, and implies (Equations (15) and (16)) that

$$\sqrt{\frac{\bar{z}A}{Q}} = \frac{SA}{Q} \quad ,$$

or (17)

$$S = \frac{Q}{A}\sqrt{\frac{\bar{z}A}{Q}} = \left(\frac{Q}{A}\bar{z}\right)^{\frac{1}{2}} \quad ,$$

indicating that the sedimentation or loss rate of phosphorus may be dependent upon a combination of surface hydraulic loading and mean depth.

Average annual values of the total phosphorus influent to a lake and the annual average surface hydraulic loading are required for the model. Generally, the lake area, mean depth, volume, total inflow, and weighted average influent phosphorus concentration should be known.

The loading graphs show only whether a lake is likely to be eutrophic or oligotrophic. The mass balance equation can be used to estimate the average annual steady-state *in situ* phosphorus concentration for different loading rates. Vollenweider's loading graphs have been used extensively to compare a number of lakes. Figure 1 shows a plot of 13 lakes in North America and compares their real trophic state to predictions of their trophic state.

The loading graphs can be used in a "predictive" sense by plotting the position of a lake under new phosphorus loading conditions. Vollenweider has done this succesfully for Lake Washington (Vollenweider, 1975).

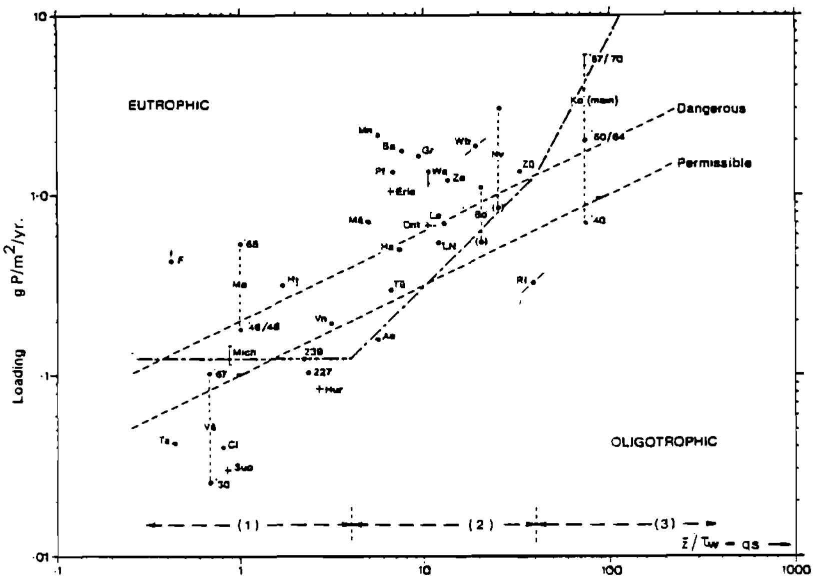

Figure 1. A plot of 13 North American lakes, comparing their real trophic state to predictions of their trophic state. The abbreviations used are: Me--Lake Mendota (e); Mn--Lake Menona (e); Wa--Lake Washington (e); Sup--Lake Superior (o); Mich--Lake Michigan (o-m); Hur--Lake Huron (o); Erie--Lake Erie (e); Ont--Lake Ontario (m); Ta--Lake Tahoe (o); Cl--Clear Lake (m); 227--ELA Lake No. 227 (o--experimentally eutrophied); 239--ELA Lake No. 239 (o); Ko--Lake Kootenay (main basin) (m-e).

SOURCE: Vollenweider (1975).

Larsen and Mercier

Larsen and Mercier (1975) presented graphs relating influent concentration to a lake's ability to assimilate phosphorus from the steady-state solution of a phosphorus mass balance model. The model was formulated in the same way as Vollenweider's.

$$\frac{dC}{dt} \approx \frac{QC_{in}}{V} - \frac{Q}{V}C - K_L C \quad , \tag{18}$$

where C is the average annual phosphorus concentration, g/m^3; C_{in} is the average annual influent phosphorus concentration, g/m^3; V is the lake volume, m^3; Q is the annual hydraulic inflow/outflow, m^3/yr; K_L is the sedimentation coefficient, per year; and t is the time, yr. It should be noted that K_L was formulated in a slightly different manner than S, $K_L = S\frac{A}{V} = \frac{S}{\bar{z}}$ (when $\bar{z}$ is mean depth).

The steady-state solution for a single, completely mixed volume is

$$C_{ss} = \frac{\frac{QC_{in}}{V}}{\frac{Q}{V} + K_L} = \frac{C_{in}Q}{Q + K_L V} = C_{in}\frac{1}{1 + K_L\frac{V}{Q}} \quad . \tag{19}$$

To estimate the value of K_L, a retention coefficient, R_p, was defined as influent minus effluent phosphorus divided by the influent value.

$$R_p = \frac{\text{mass in} - \text{mass out}}{\text{mass in}} \quad ,$$

$$R_p = \frac{C_{in}Q - CQ}{C_{in}Q} \quad , \qquad \text{since } C = C_{in}\frac{1}{1 + K_L\frac{V}{Q}} \quad ,$$

$$R_p = \frac{C_{in}Q - \frac{C_{in}Q}{1 + K_L\frac{V}{Q}}}{C_{in}Q} \quad , \tag{20}$$

$$R_p = \frac{K_L}{\frac{Q}{V} + K_L} \quad .$$

It was recommended that influent phosphorus concentration should be plotted against the retention coefficient, R_p, to determine a lake's trophic state. The influent phosphorus concentration and the lake phosphorus retention coefficient are essential input parameters. This procedure will place a lake on the graph in order to estimate its trophic state.

Dillon

Dillon (1974) proposed the same mass balance as those used by Vollenweider and Larsen and Mercier

$$\frac{dC}{dt} = \frac{QC_{in}}{V} - K_D C - \frac{Q}{V}C \quad , \tag{21}$$

where all terms have been defined previously, and K_D is equal to K_L of Larsen and Mercier. The time variable and the steady-state solution to the mass balance equation are provided. The concentration as a function of time is given by

$$C_{(t)} = \frac{QC_{in}}{Q + K_D V}\left(1 - e^{-\frac{Q+K_D V}{V} t}\right) + C_o e^{-\frac{Q+K_D V}{V} t} \quad . \tag{22}$$

The steady-state or infinite time solution is

$$C_{ss} = C_{in} \frac{Q}{Q + K_D V} \quad . \tag{23}$$

Dillon proposed methods for estimating the necessary parameter values, including hydraulic loading, phosphorus loading, and phosphorus sedimentation rate. He suggested that the phosphorus retention could be computed as

$$R = 0.426 \exp(-0.271\ q_s)$$

$$+ 0.574 \exp(-0.00949\ q_s) \quad , \tag{24}$$

where q_s is the surface hydraulic loading rate in m/yr. Dillon and

Rigler (1975) later suggested that R_p could be estimated in the same manner as previously discussed for Vollenweider's and Larsen and Mercier's models. Dillon also pointed out the importance of response time in an analysis of nutrient budgets. Rather than model time-dependent response, he noted that the "half life" of the change in concentration (time required to change half way from old concentration to a new equilibrium concentration in response to changed loading) can be computed from Equation (23) to be

$$t_{1/2} = \frac{0.69}{V/Q + K_D} \quad . \tag{25}$$

The annual phosphorus loading, lake dimensions, and hydraulic flow rates are essential input parameters. In addition, the sedimentation coefficient, K_D, must be estimated. Dillon's procedure allows computation of a permissable phosphorus load to a lake. The amount depends upon selected permissable in-lake concentrations. When loading rates are given, the same mass balance model can be used to predict phosphorus concentrations in the lake.

Lorenzen et al.

Lorenzen et al. (1976) developed a phosphorus budget model that considered total phosphorus only, but it included uptake and release from the sediments, as well as permanent sediment storage. A mass balance on both sediment and water phosphorus concentration yields the following coupled differential equations.

$$\frac{dC}{dt} = \frac{QC_{in}}{V} + \frac{K_2 A C_s}{V} - \frac{K_1 A C}{V} - \frac{CQ}{V} \quad , \tag{26}$$

$$\frac{dC_s}{dt} = \frac{K_1 A C}{V_s} - \frac{K_2 A C_s}{V_s} - \frac{K_1 K_3 A C}{V_s} \quad , \tag{27}$$

where c_{in} is the influent phosphorus concentration, g/m^3; C is the average annual total phosphorus concentration in water column, g/m^3; C_s is the total exchangeable phosphorus concentration in the sediments, g/m^3; V is the lake volume, m^3; V_s is the sediment volume, m^3; A is the lake surface area, m^2, which is equal to the sediment area, m^2; Q is the annual inflow/outflow, m^3/yr; K_1 is the specific rate of phosphorus transfer to the sediments, m/yr; K_2 is the specific rate of phosphorus transfer from the sediments, m/yr; and K_3 is the fraction of total phosphorus input to sediment that is unavailable for the exchange process. These equations were solved both numerically and analytically to give the average annual phosphorus concentrations as a function of time in the water and in the sediment.

The steady-state solution is equivalent to the simpler models of Vollenweider, Larsen and Mercier, and Dillon, with a slightly different interpretation of the sedimentation rate constant. The model presented in Lorenzen et al. gives a steady-state phosphorus concentration in the water column equal to

$$C_{ss} = \frac{C_{in}Q}{Q + K_1K_3A} = C_{in}\,\frac{1}{1 + K_1K_3\,\frac{A}{Q}} \quad . \tag{28}$$

Here K_1K_3 equals Vollenweider's S, Larsen and Mercier's K_L times $\bar{z}$, and Dillon's K_D times $\bar{z}$. It is interesting to note that Lorenzen et al. showed the ratio of steady-state water to total available sediment phosphorus by

$$\frac{C_{ss}}{C_{s,ss}} = \frac{K_2}{K_1}\,\frac{1}{(1 - K_3)} \quad . \tag{29}$$

The time-dependent solution is considerably more complex, and a "half life" cannot be computed independently of initial sediment and water concentration.

For the dynamic model application, total annual phosphorus loading, lake dimensions, and hydraulic flows must be known. In addition, values for the three rate constants--K_1, K_2, K_3--must be provided, and the initial water and sediment phosphorus concentrations must be known. The steady-state solution requires the same information as the models of Vollenweider, Dillon, and Larsen and Mercier. The model computes annual average concentrations of total phosphorus in the water column and the sediments.

Imboden

Imboden (1974) developed a phosphorus budget model for lakes that considers two forms of phosphorus (dissolved and particulate), two layers (epilimnion and hypolimnion), and exchange with the sediments. The sediment phosphorus concentration is not modeled, but rates of oxygen consumption are computed.

The model formulation results in the following four coupled differential equations.

$$\frac{dC_{de}}{dt} = -C_{de}\left(\frac{Q}{Z_eA} + E + \alpha\right) + C_{dh}E + C_{pe}M_e + \frac{C_{d,in}Q_{in}}{Z_eA} \quad , \tag{30}$$

$$\frac{dC_{dh}}{dt} = C_{de}\left(E\,\frac{V_e}{V_h}\right) - C_{dh}E\left(\frac{V_e}{V_h}\right) + C_{ph}M_h + \frac{S}{Z_h} \quad , \tag{31}$$

$$\frac{dC_{pe}}{dt} = \alpha C_{de} - C_{pe}\left(\frac{Q}{AZ_e} + E + M_e + \frac{K_I}{Z_e}\right) + C_{ph}E + \frac{C_{p,in}Q_{in}}{Z_e} \quad , \tag{32}$$

$$\frac{dC_{ph}}{dt} = C_{pe}\left(E\,\frac{V_e}{V_h} + \frac{S}{Z_h}\right) - C_{ph}\left(E\,\frac{V_e}{V_h} + \frac{K_I}{Z_h} + M_h\right) \quad , \tag{33}$$

where C is the concentration , g/m^3; Q is the flow, m^3/day; Z is the depth, m; V is the volume, m^3; A is the area, m^2; E is the exchange rate, per day; α is the phosphorus uptake rate, per day; M is the mineralization rate, per day; S is the exchange with sediment, mg/m^2/day; K_I is the sedimentation rate, mineraliz./day; and t is the time, days; and for the subscripts, d is dissolved, p is particulate, e is epilimnion, h is hypolimnion, and in is influent.

Total oxygen consumption rate in the hypolimnion was computed to be

$$\frac{d}{dt}[O_z] = -140C^{\circ}_{pe}\left[\frac{K_I}{Z_h} + \frac{EM_h}{E + \frac{K_I}{Z_e} + \frac{V_h}{V_e}M_h}\right] \quad , \tag{34}$$

where C°_{pe} is the steady-state epilimnion particulate phosphorus concentration.

In addition to phosphorus loading and hydraulic and morphometric data, this model requires the specification of thermocline depth and time of occurrence, sedimentation rate for particulate phosphorus, exchange rates between epilimnion and hypolimnion, as well as hypolimnion and sediment, and phosphorus uptake rate by algae. The model should compute daily concentrations of particulate and dissolved phosphorus in both the epilimnion and hypolimnion of a lake. Imboden (1974) used these formulations to compute a theoretical phosphorus retention for thirteen lakes. The computed values were compared with observations as shown in Table 2.

Snodgrass and O'Melia

Snodgrass and O'Melia (1975) developed a phosphorus budget model for lakes that is essentially the same as Imboden's, except that sediment phosphorus release is not considered. Two forms of phosphorus (soluble orthophosphorus and particulate phosphorus) were considered. Two layers, (epilimnion and hypolimnion) were included, and loss of particulate phosphorus to the sediments was

Table 2. Comparison of theoretical and measured phosphorus retention.

Lake and abbreviation	qs (m/day)	z (m)	Retention factor (%)	
			theory	measured
Aegerisee A	0.019	50	67	67
Baldeggersee B	0.015	34	79	61
Lake Constance Bo	0.058	90	27	65
Greifensee G	0.037	19	72	62
Hallwilersee H	0.019	21	83	36
Lake Geneva L	0.036	154	26	20
Pfäffikersee P	0.033	18	74	77
Sempachersee	0.0074	46	94	--
Türlersee T	0.018	14	84	80
Zürichsee Z	0.098	44	31	25
Lake Tahoe Ta	0.001	300	86	93
Vänern V	0.001	36	99	--
Zellersee Ze	0.038	37	65	--

modeled. Periods of winter circulation and summer stratification were considered separately.

For the summer condition, four coupled, linear differential equations describe the mass balances of both forms of phosphorus in the epilimnion and hypolimnion. For orthophosphorus (OP) in the epilimnion,

$$V_e \frac{d(OP)_e}{dt} = \Sigma\, Q_j(OP)_j - Q(OP)_e - p_e V_e (OP)_e \quad ,$$

$$+ \frac{k_{th}}{\bar{z}_{th}} A_{th}(OP)_h - \frac{k_{th}}{\bar{z}_{th}} A_{th}(OP)_e \quad . \qquad (35)$$

For particulate phosphorus (PP) in the epilimnion,

$$V_e \frac{d(PP)_e}{dt} = \Sigma\, Q_j(PP)_j - Q(PP)_e + p_e V_e (OP)_e$$

$$- g_e A_{th}(PP)_e + \frac{k_{th}}{\bar{z}_{th}} A_{th}(PP)_e \quad . \qquad (36)$$

For orthophosphorus in the hypolimnion,

$$V_h \frac{d(OP)_h}{dt} = r_h V_h (PP)_h + \frac{k_{th}}{\bar{z}_{th}} A_{th}(OP)_e$$

$$- \frac{k_{th}}{\bar{z}_{th}} A_{th}(OP)_h \quad . \qquad (37)$$

For particulate phosphorus in the hypolimnion,

$$V_h \frac{d(PP)_h}{dt} = g_e A_{th}(PP)_e - g_h A_s (PP)_h - r_h V_h (PP)_h$$

$$+ \frac{k_{th}}{\bar{z}_{th}} A_{th}(PP)_e - \frac{k_{th}}{\bar{z}_{th}} A_{th}(PP)_h \quad . \qquad (38)$$

The subscripts e and h refer to the epilimnion and hypolimnion; th and s denote the thermocline region and the sediment/water interface; OP and PP refer to the concentrations of orthophosphate and particulate phosphorus; p and r are rate coefficients for net production and decomposition; k is a vertical exchange coefficient, L^2T^{-1}, that includes the effects of molecular and turbulent diffusion internal waves, erosion of the hypolimnion and other fluid processes on the transfer of heat and materials across the thermocline; $\bar{Z}$ is a mean depth; V is the volume; A is an interfacial area; Q_j is a land-based volumetric rate of inflow of water; Q is the volumetric rate of lake discharge; and g is an effective settling velocity.

For the winter model, two linear differential equations are obtained from mass balances of the two phosphorus compartments. For orthophosphorus throughout the lake,

$$V \frac{d(OP)}{dt} = \Sigma\, Q_j(OP)_j - Q(OP) - p_{eu}V_{eu}(OP) + rV(PP) \quad . \tag{39}$$

For particulate phosphorus throughout the lake,

$$V \frac{d(PP)}{dt} = \Sigma\, Q_j(PP)_j - Q(PP) + p_{eu}V_{eu}(OP) - rV(PP) - gA_s(PP) \quad , \tag{40}$$

where the subscript eu denotes the euphotic zone.

These models are joined by boundary conditions at the fall overturn and the spring stratification. At the fall overturn,

$$(OP) \text{ just after mixing} = \frac{(OP)_e V_e + (OP)_h V_h}{V} \quad , \tag{41}$$

$$(PP) \text{ just after mixing} = \frac{(PP)_e V_e + (PP)_h V_h}{V} \quad . \tag{42}$$

At the beginning of the spring stratification, the concentrations of particulate phosphorus in the epilimnion and the hypolimnion are set equal to the concentration of particulate phosphorus throughout the lake at the end of the winter circulation: $(PP)_e = (PP)_h = (PP)$. Similarly, $(OP)_e = (OP)_h = (OP)$.

Just as Imboden's model, this model requires specification of loading rates, lake geometry, hydraulic conditions, production and respiration rates, sedimentation rates, and exchange coefficients. Variable parameters must be specified on a daily basis. The model should compute daily values for concentrations of orthophosphorus and particulate phosphorus in both the epilimnion and hypolimnion. Snodgrass and O'Melia used this model to compute spring total phosphorus concentrations in 11 lakes. A comparison between computed and observed values is shown in Table 3.

RECOMMENDATIONS

The selection of the specific procedures to be used as part of the evaluation of the impact of phosphorus control options is constrained by a number of factors. The large number of lakes to be used as a data base requires a relatively simple approach. The time scale of interest requires a method that can be applied to predict years into the future. The type of control options and specific effects on loading rates requires a tool that considers the direct result of the control.

Table 3. Predicted and observed spring total phosphorus concentrations.

Lake	Predicted spring total Phosp. mg/liter	Observed spring total Phosp. mg/liter
Aegerisee	8	8
Türlersee	18	13
Zürichsee	28	32
Bodensee	30	35
Sempachersee	48	35-37
Ontario	25	23-27
Tahoe	2	2-4
Kalamalka	23	20
Okanagan	24	30
Shaha	65	59
Osoyoos (N. Basin)	69	59

Phosphorus Loading

The best procedures for computing new phosphorus loading rates resulting from various control options will depend very much on the type of control option specified. Conversely, the types of control options that can be analyzed depend greatly upon the methods and data available. For example, the effect of a change from agricultural to urban land use can be evaluated by using the phosphorus loading data compiled by Omernik. However, this can be done only for those lake basins that have available present land-use data. For point source control, an option such as the elimination of phosphorus detergents could be analyzed either by reducing municipal treatment plant loads by a fixed percentage, or by specifying a new effluent concentration. One of the difficulties involved in specifying effluent concentrations is that the "compendium" data base does not include flow data. Flow data are available from the STORET data base but would have to be supplied.

In general, the procedures described by Porcella et al., are recommended for evaluation of point source control options and the data base reported by Omernik should be used for nonpoint source analysis. A further analysis of Omernik's data base may be necessary, in order to arrive at a suitable method for predicting nonpoint source loading.

Lake Response

Owing to the lack of data and tested models, it is recommended that the single equation mass balance model be used for evaluating lake responses. Such a model will be unable to account for sediment release of stored nutrients during a transient recovery period. However, the data base contains no information on sediment phosphorus concentrations. A more complex model would require estimation of rate constants and sediment phosphorus concentrations.

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THEORETICAL INVESTIGATIONS INTO THE DEVELOPMENT OF AN INTENSIVELY USED SHALLOW LAKE WITH A RIVER FLOWING INTO IT

K. Bauer

The shallow lake considered in this paper is the Grosser Müggelsee, which forms a part of the water system subordinated to the management district of the capital of the GDR, Berlin. In such a metropolis it is essential to guarantee water supply and recreational areas; thus, development of this shallow lake is envisaged, in order to satisfy future demand (see Figure 1).

Since 1895, the Grosser Müggelsee has been the subject of many limnological investigations (e.g., Müller 1895, Lemmermann 1903, Bursche 1953, Barthelmes 1959, Schlüter 1966, Warnke 1970, Bünger 1975, Cramer 1976, and Bauer and Warnke 1977). The investigations were based on daily and weekly data of ecosystem components and chemical pollutants, measured over a period of 32 years. Changes in the composition of phytoplankton species were not considered.

STATISTICAL ANALYSIS

By using the correlation theory, which forms part of the stochastic processes theory, to evaluate the available data, some interesting effects have become apparent. Some problems and findings related to the data evaluation follow. The statistical analysis of the water system under consideration was based on the

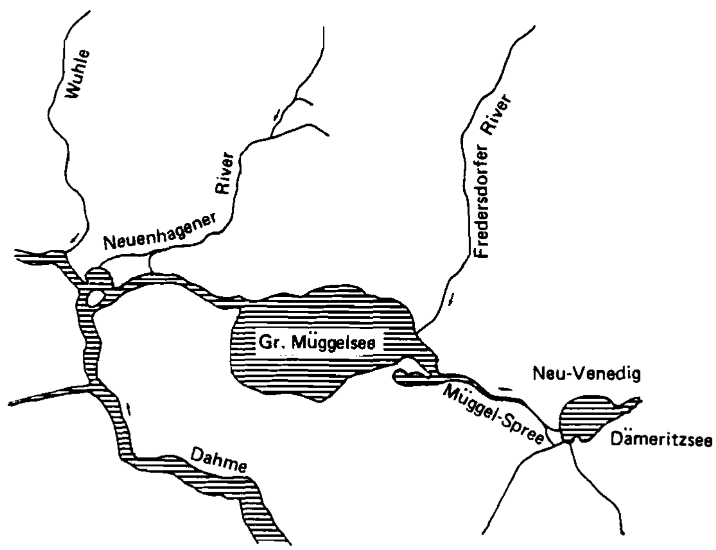

Figure 1. The position of the Grosser Müggelsee in the water system of Berlin, GDR.

following ecosystem variables (the observation periods are in parentheses):

Silicate (1 year)
Water temperature (32 years)
Nitrate (1 year)
Global radiation (1 year)
Chlorophyll-a (1 year)
Orthophosphate (1 year)
Ammonia (1 year)
Oxygen content (32 years)
Nitrite (1 year)
Carbonate hardness (32 years)
Chloride (32 years)
Suspended matter (32 years)
Potassium permanganate consumption (32 years)

Three series of data were obtained from the water system under consideration. Since a part of these data series was complex, we had to find a new procedure for processing the series. Computing operations were performed on a statistical basis. Selected results from an evaluation of the ecosystem components are displayed in Table 1.

Table 1. An evaluation of the ecosystem's components, selected results.

Component	Mean value		Variance	Residual variance	RVA_{NF} [a] %	1st zero points ACF [b] days
Silicate	8.3	mg/liter	27.4	0.87	97	20
Water temperature	11.76	^{o}C	53.5	2.25	96	9
Nitrate	5.8	mg/liter	6.2	0.92	85	7
Global radiation	1,489.8	cal/m^2	9.4×10^5	1.40×10^5	85	7
Chlorophyll-a	91.9	μg/liter	8,632.0	1,652.0	81	15
Orthophosphate	0.96	mg/liter	0.11	0.025	77	7
Ammonia	1.5	mg/liter	0.51	0.136	73	15
Nitrite	0.22	mg/liter	0.0064	0.0018	72	10
Oxygen saturation	85.4	percent	851.3	331.6	61	8

[a] RVA_{NF} is the relative fraction of the fundamental low-frequency variation in the variance.
[b] ACF is the autocorrelation function.

All components of the ecosystem (see Table 1) show low-frequency 1-year variations. With regard to temperature more than 96 percent of the total variance occurs in the low-frequency range. For all remaining ecosystem components the portion of the relative variance in the fundamental low-frequency variation amounts to more than 50 percent. In the case of a slowly flowing nutrient-rich watercourse the annual variation in oxygen content may be only 61 percent. This can be attributed to the influence of the temperature on the oxygen saturation concentration. The remaining 39 percent might be largely due to the influence of biogenic aeration. If the annual variations of the ecosystem components under consideration are separated, the correlation functions of the stochastic behavior give zero values in the interval between lags τ is 7 days and τ is 20 days. Silicate, chlorophyll-a, and ammonia prove to be stochastically inert components.

For the longer time series of variables--water temperature, oxygen, suspended matter, chloride, chemical oxygen demand, and carbonate hardness--the following assessment can be made on the basis of the monthly means during the period from 1945-77 (see Figure 2).

Temperature, Oxygen, Suspended Matter

By comparing the monthly trends of water temperature and oxygen, it becomes obvious that meteorological factors have a considerable influence upon variations in the oxygen content. Until recently the annual oxygen variation between spring and summer had been rising. The increase in suspended matter is clearly indicated by a significant growth of plankton, by an earlier start in algal production and by the more frequent appearance of two peaks in algal growth, the first peak appearing in spring and the second in summer.

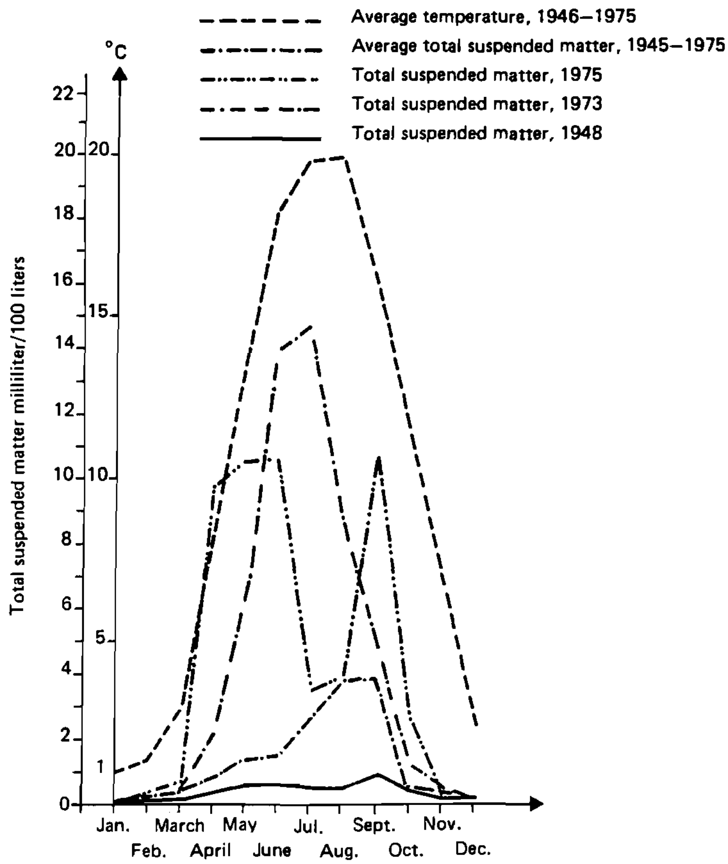

Figure 2. Yearly variations: suspended matter in the Grosser Müggelsee.

Chloride, Chemical Oxygen Demand (oxygen consumed in parts per million (ppm) potassium permanganate), Carbonate Hardness

The chloride ion is regarded as the indicator ion. The increased amounts of chloride in the water point to a greater influence of industry upon the catchment area (e.g., use of desalination plants to generate electricity), which is apparently contrary to the prevailing negative trend in the chemical oxygen demand as measured by potassium permanganate. The decrease in potassium permanganate values indicates a reduction in easily oxidizable organic matter in the water, whereas the findings of special investigations show that matter not easily oxidized is increasing. With regard to carbonate hardness a yearly variation can occasionally be observed, but no general trend is apparent. Unfortunately, such long and reliable series of measurements of nutrient and organic matter are unavailable. For these important ecosystem components we have only a representative series of daily values for a 1-year observation period. It may be concluded that the increase in nutrients in the water of the catchment area, brought about by a systematic increase in industrial and agricultural activity, may be the cause of the increased algal production in the Grosser Müggelsee.

Given the available data on suspended matter, the first goal with respect to sanitation should be to obtain an increase in the potable water supply from the Grosser Müggelsee. In order to achieve this aim more attention must be given to the difficulties involved in water treatment; difficulties caused primarily by the presence of suspended matter in the water. It would be desirable to reduce by one-third the amount of suspended matter, but if this is not possible at least the plankton growth, as shown by the long-term time series, should be curbed. This might be possible if the overall nutrient input into the catchment area were to be reduced.

In the GDR the quality of water bodies has been improved. This has been achieved by carefully judging when to use nutrients in agriculture and forestry, and by preventing the wastewater treatment plants from releasing their discharges into nearby rivers and lakes.

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EUTROPHICATION MODELING EFFORTS FOR LAKE BALATON

P. Csáki, J. Fischer, L. Hajdu, and G. Jolánkai

Lake Balaton has a surface area of almost 600 square kilometers. Along its southern shore it is shallow for approximately 500 meters, and from May until the end of September the water is warm. These two features contribute towards making this lake a popular recreational center. The principal characteristics of the lake are included in Table 1 (see also Figure 1).

Table 1. Morphological and hydrological characteristics of Lake Balaton.

Morphological and hydrological features	Unit of measurement
Surface area	593.00 km^2
Average width	7.68 km
Average depth	3.14 km
Length	77.80 km
Volume	1,861.00 x $10^6 m^3$
Drainage basin area	5,774.50 km^2
Average water-level fluctuation	48.00 cm
Long-term average annual inflow in water column	~954.00 mm
Long-term average annual outflow	~681.00 mm

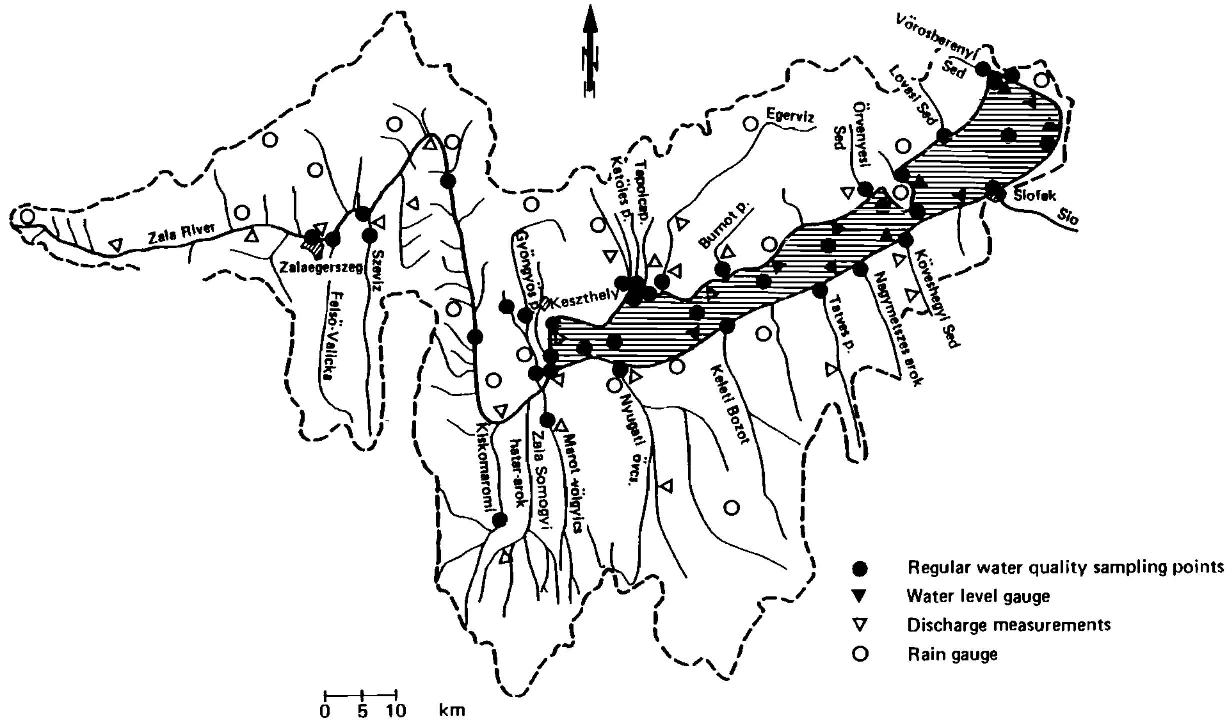

Figure 1. Lake Balaton and the catchment area.

About 75 percent of the Lake's total inflow comes from the discharge of the River Zala into the farthest southwestern end of the lake. The outflow at the farthest northeastern end is regulated by a gate that discharges into the Sio' Channel (see Figure 1). Many conditions of the lake, including detention times and water quality, can be explained by this specific inflow/outflow feature.

FEATURES AFFECTING POLLUTION LEVELS IN LAKE BALATON

Along the southern shore of the lake there is comparatively little industry, that which exists consists mainly of agriculture and related services--e.g., wine processing. Along the northern shore there are boatyards and chemical works. Of the total land area in the Balaton watershed, 85.2 percent is used for agriculture and 14.2 percent for forestry; the remaining 0.6 percent is accounted for by other activities. Most types of industrial pollution can be found in the catchment area of the River Zala.

The size of the population in the Balaton watershed is another factor that indirectly affects the water quality of the lake. The permanent population in the recreational region is 120,000; however, during the peak tourist season in summer, the population is estimated at 600,000. An adequate water supply is essential to such a region, and although present capacity is approximately 120,000 m^3/day, it is being increased rapidly to cater for the needs of the area. A regional canalization network is also being developed simultaneous with the construction of sewage treatment plants. At present many settlements use septic tanks. When the regional network is completed all sewage will be treated before being conveyed from the Balaton Basin into the lake.

The amount of traffic traveling along the shoreline is the third factor contributing to pollution levels in the lake. The level of railway transport remains constant, but private car traffic has rapidly increased. During the last 10 years this increase has been calculated at between five and six times the previous level.

EUTROPHICATION PROBLEMS IN LAKE BALATON

In current lymnological textbooks, Lake Balaton has been mentioned as a typical example of a eutrophic lake; this is certainly an exaggeration. The water of this lake is extremely rich in calcium. Annual biogenic calcium precipitation is estimated at 80,000 tons, and the calcium carbonate content of the bottom sediment is 50 to 60 percent. Because the fine calcium grains absorb phosphorus effectively, eutrophication is diminishing. However, even by this efficient defensive mechanism, the process cannot be stopped completely; this is because the external phosphorus load has most probably greatly increased. In the phosphorus absorption mechanism, the aerobic bottom sediment usually plays the key role, but in Lake Balaton it is the high level of pH in the water (8.4) that is important for the process. Occasional acidification, biologically realistic down to the neutral value of 7.0, causes the apatite in the sediment to dissolve, which in turn results in the release of phosphorus.

Originally the major influent, the River Zala, flowed through the shallow marsh Kisbalaton (Little Balaton). This served as a natural tertiary stage for the treatment of waste material, eliminating the bulk of biogenic elements and transported abioseston from the water. This natural filter was closed off by the construction of a canal, and as a result there has been a gradual buildup of deposits in the western part of the lake. Concentrated nutrient loading from agricultural and communal sources has greatly accelerated the process of eutrophication.

Recently, unusual phytoplankton species have been observed, and the natural diversity of the algae has been curbed by the dominance of one particular species (Bartha, 1975; Hadju, 1976). In the 1930s it was unusual to find an area of several square meters of algal bloom in Lake Balaton. This is not the case now because over the last few decades it has spread rapidly; as yet, it has not become a serious problem. Biomass and phytoplankton production have increased, and the latter has taken place more rapidly, as indicated by observations of algal species and primary production measurements, for example, see Herodek and Olah, 1974; Herodek and Tamas, 1975. Eutrophication in the lake suddenly became a problem in the 1970s. Production reached a peak between 1970 and 1973, when in the western part, the Keszthely Bay, the theoretical maximum production level was attained, and when along the eastern shore a new species of algae occurred in dense clumps, which eventually spread to other areas of the lake.

The introduction into the lake of the zebra mussel *(Dreissena polymorpha)* and the eel *(Anguilla anguilla)*, by natural and artificial means respectively, has had an unfavorable effect on the lake's indigenous fauna; this is demonstrated, for example, by the considerable decrease in the mollusc *(Anodonta cygnea)* and the crab *(Potamobius leptodactylus)* populations.

In 1965 and 1975 there were two large fish kills, the circumstances of which have not yet been clarified by experts. However, it is considered probable that the first case, in 1965, was a result of pesticide intoxication from agricultural land runoff into the lake, and the second, in 1975, was due to a lack of oxygen in the water caused by the increased growth of phytoplankton.

The recreational value of the lake has not yet been significantly impaired. Clumps of seaweed occur sporadically, and in the western bay this may deter some people from bathing. The shallow water is aerated to a significantly great depth to ensure that the bottom sediment is always aerobic, containing little organic matter.

The dangers of eutrophication in Lake Balaton were recognized at a relatively early stage, before serious biological problems had occurred. The credit for this is due primarily to the Biological Research Institute at Balaton and the Budapest Research Center for Water Resource Development.

THE CAUSES OF EUTROPHICATION

Most experts on Lake Balaton's eutrophication problem agree that the lake is phosphorus-limited--i.e., phosphorus input into the lake is the cause of the eutrophication process. While this assumption is true in general, one specific condition should always be remembered. Since there is almost total, instantaneous precipitation of phosphorus in the water, which contains a high calcium carbonate content, the main direct source of phosphorus can be attributed to the exchange process across the water/sediment interface. Therefore a long period of phosphorus accumulation in the sediment was necessary before equilibrium concentration in the water began to rise, with a subsequent increase in trophic levels. Although this retarding effect has served the interests of water quality, it could also delay the effect of phosphorus removal from the sources of pollution. The multiplicity of phosphorus sources should also be considered in a discussion of the causes of eutrophication in Lake Balaton. These sources consist of waste-water from communal and industrial origin; runoff from agricultural land; urban runoff from the densely populated areas; and direct precipitation on the water surface. Although there recently have been many attempts to determine the exact contribution of these sources to pollution levels in the lake, only approximate, and sometimes contradictory, estimates can be given due to the lack of detailed information and regular, longstanding input records. Nevertheless, we estimate the contribution of the different pollution sources as follows (expressed as a percentage of the total input): sewage, 30 percent; nonpoint (runoff), 40 percent; urban runoff, 10 percent; precipitation, 20 percent. A recent calculation of phosphorus mass balances in the lake, based on available data, is given in Table 2. To the total inflow of phosphorus a certain nonpoint source contribution must be added, although some runoff effects may also be included in the values shown in Table 2.

Table 2. Phosphorus mass balances in Lake Balaton.

Sources of pollution	Total phosphorus in kg/day
Tributaries on the northern shore	76.36
Tributaries on the southern shore	62.8
River Zala	430.54
Direct discharge of sewage treatment plant onto the shore	106.96
Total inflow	682.06

The unobserved rainfall runoff contribution, according to field studies, could be of the same order of magnitude as the total sum given in Table 2. Taking into account these estimates the total load in the lake can be calculated at about 2,000 kg/day. Since outflow is almost negligible (about 30 kg/day) there is a considerable accumulation of phosphorus 1.26 $g/m^2/yr$. Only the dissolved reactive form of phosphorus, PO_4-P, can be absorbed by the algae, being the first (and probably most significant) element in the food chain. Unfortunately there are no reliable measurements of PO_4-P inputs available, with the exception of those for the River Zala.

MODELING EFFORTS

The Objective

The principal objective of the modeling efforts is to develop a eutrophication model for the lake that fulfills the following conditions. The model should serve as a control tool, describing the eutrophication process with reasonable accuracy; it should allow a description of nutrient inputs and nonpoint sources; and it should be able to operate within the constraints of the available data base. For more details of the first stage of this modeling approach see the paper by G. Jolánkai in these Proceedings. At a later stage the model should be extended to include all the determinative trophic state parameters. However, to satisfy all these requirements it will be necessary to increase modeling and monitoring activity over the long term. For the present, efforts should concentrate on simpler forms of eutrophication modeling.

Data Availability

Reliable historical records on the hydrological and meteorological conditions in the Balaton watershed are available, but they do not include inflow measurements (discharge measurement structures are installed only on some of the principal tributaries). However, recently discharge measurements for the River Zala, the main source of inflow, have been taken more frequently lately. Stations for the observation of hydrological, meteorological, and water quality conditions are shown in Figure 1.

The frequency of water quality sampling on the tributaries varies between 3 and 56 measurements, annually, and it is increasing. During the previous year samples of the River Zala were taken daily. At the points where samples are taken the most frequently, routine analysis is carried out for 46 water quality components. The frequency of water quality sampling in the lake is also increasing; at present about 15 samples are taken annually. Detailed analyses of traditional water quality components, phosphorus and nitrogen forms and algal chlorophyll are carried out. All the data on water quality in the lake have been gathered to form a data bank, which at present is being installed on computer, and the organization of the enviromental monitoring system for the region is in progress.

Regular influent sampling is not yet sufficiently frequent to provide reliable data on nutrient input 4-6 measurements annually. Extensive objective-oriented measurements were organized previously and are also being carried out at present.

Of all the biological data characterizing the lake algal records are the most important; they can be interpreted as a direct measure of eutrophication. In the early stages of research on Lake Balaton, up to 1950, only qualitative data were collected. More recently, 1950-1960, quantitative work has been carried out with the aid of a Kolkwitz chamber, which does not allow a comparison with the data of the Utermöhl microscope countings, as is usual today. For biological purposes water quality samples are being taken about six times annually from five cross-sections of Balaton. The number of alga individuals and their volumetric biomass have been calculated by using three-dimensional models, see Table 3. It should be noted that the maximum, for Keszthely Bay, is about 10 mg/liter. The collection of volumetric biomass data is a laborious activity, and therefore there is little data available. An estimation of that value by chlorophyll-a quantity is more usual, because measurements are much easier to take.

Primary production samples by the C14 method have been made since 1962, and nitrogen fixing of blue-green algae was first analyzed in 1977.

Table 3. A data series of three years' averages.

	Apr	May	June	July	Aug	Sept	Oct	Nov
biomass mg/liter	0.36	0.46	1.21	2.14	4.11	1.84	0.48	0.18

The process of eutrophication in the lake can be followed not only in time, but also in space: its direction is West-East. Different types of diversity calculations and cluster analyses have been performed in order to gain more information about the process. Regular records of algal stands have also been made, such as those of Kárpáti and Kárpáti (1975). Bacteriological life in Lake Balaton has been well documented, and the bacterium count on membrane filters and its annual changes are well known. However, records on zooplankton populations are inconsistent and irregular, although extensive research has been carried out on the nutrition and reproduction of an important zooplankton species, *Eudiaptomus gracilis* (Zánkai and Ponyi, 1975). Total fish biomass in the lake is estimated at approximately 6,000 tons with an annual production of about 3,000 tons. The total quantity of fish caught by anglers is about 23 tons.

Reliable historic records on basic parameters of water chemistry, principal cations, conductivity, and the like are available, but these data have little relevance to ecological modeling. Data obtained by using modern analytical techniques, such as data on various forms of phosphorus, have only been available in the last 6 years.

The Modeling Approach

Nonpoint source pollution, in particular runoff from agricultural land, makes a significant contribution to the pollutant input into Lake Balaton. In the past modeling attempts to describe this phenomenon have been made. In a representative watershed of the lake, extensive field measurements have been taken, in order to investigate the effect of runoff on the lake's plant nutrient input. Weekly sampling and discharge measurements were organized with increased sampling frequency during flood events. Rainfall data and data for agricultural chemicals used in the area were also collected.

The modeling of base flow mass flux conditions has been solved by constructing mass flux profiles corresponding to different probabilities of occurrence for each of the eight parameters investigated. With an extension of the available hydrobiological model, pollutant runoff events have been simulated by using the correlations between pollutant mass fluxes and flood wave volumes (Jolánkai, 1977).

The data from this extensive field study have also been used in a more sophisticated stochastic approach (Duckstein et al., 1976) that forecasts runoff events and also sediment yield events from rainfall, making use of the available data on dissolved phosphorus and fixed phosphorus. The probable density functions of phosphorus loading from nonpoint agricultural sources is estimated.

There are some promising results with a eutrophication model of the lake's most endangered part. The model, in its present form, is a phosphorus/algae type that takes the exchange process between sediment and water into consideration. Its main feature is that it considers time variable reaction coefficients, and uses a maximum likelihood approach as a tool for parameter identification. An extension of this model is being developed and will include:

- The prediction of inflow from the main tributary, and a joint model of dissolved reactive phosphorus input from this source
- Water-level fluctuation and outflow (joint model with water quality components' outflow)
- Sediment phosphorus precipitation into and uptake from the sediment including wind effects
- Algal growth (light, temperature, and flood limitation)
- Other aquatic forms of life in the food chain (not yet fully agreed upon, dependent upon data availability)

In its final form the model could be an effective tool for management strategies.

Trial runs of various models have been carried out, so that the most appropriate set of ecosystem state variables may be selected. The Balaton Ecosystem Model 1 (BEM-1) seems to provide one of the most suitable initial approaches. It consists of the following state variables: inorganic nutrients in the water--phosphorus, nitrogen; organic matter in the water: dissolved and particulate (DOM and POM); and decomposers, algae, zooplankton, fish. The dynamics of these state variables are described by a differential equation system based on nutrient uptake, release (respiration, excretion, defecation), and mortality. The main features of the model are the following:

-- The expressions for most of the process kinetics are proportional to the biomass of the corresponding living components.
-- The proportionality factors consist of constant maximum rates and limiting factors.
-- The limiting factor for the temperature takes the form of Ve^{1-V} occurring in all process rate expressions.
-- The limiting factor for phosphorus, nitrogen, and light is based on the Michaelis-Menten law (for light it is averaged according to depth) limiting the primary production of the algae.
-- The food chain is expressed in terms of uptake and grazing rates: the food of algae--phosphorus, nitrogen; the food of decomposers--DOM; the food of zooplankton--algae, decomposer, POM; the food of fish--decomposer, zooplankton, POM.
-- The total mass (water + organic matter) of the phosphorus and nitrogen is constant wherever the phosphorus and nitrogen loading is zero. The carbon/nitrogen/phosphorus ratio in all forms of organic matter is constant. These requirements are realized by certain constraints on the parameters.

The next task is to incorporate the models of inorganic nutrient loading and the water/sediment exchange of the phosphorus.

In conclusion, extensive data analyses and modeling activity on the hydrology of the lake have been performed in the past.

Even a brief interpretation of this data would far exceed the scope of this paper, but, to summarize, the principal "modeling" problem is that of selecting one appropriate approach from the many that are available.

FUTURE TASKS

Despite the availability of many research results and the wide-ranging, but unconnected, data bases, the most difficult problem in the construction of a realistic management model is still the lack of data. In order to solve this problem, more data on the extent and sources of pollution need to be gathered, and certain modeling features have to be improved.

First of all this involves taking more frequent measurements along the tributaries of Lake Balaton for discharge and water quality. In order to determine the effect of seasonal variation, more frequent measurements of the effluent discharges of sewage treatment plants are also necessary. Special field studies could explore the load runoff events in some representative areas, and could gather further information on the effects of fertilizers and pesticides. In analyzing the water quality and biological features of the lake itself, more data (both in time and space) on various aquatic forms of life and a definition of the most dominant eutrophication state variables would be required. A useful contribution to solving the problem could be provided by literature, field, and laboratory research into the growth equations of various aquatic forms of life; this would include gathering information on assumed constant values, followed by an analysis of this information.

Secondly, certain aspects of modeling need to be considered. This would involve constructing a submodel of the hydrological features of the lake that would be compatible with the ecosystem model. In order to allow for continuous extension and refinement, the submodels--ecosystem, eutrophication, and input--would require calibration, and the development of up-to-date parameter estimation techniques. Control strategies should be considered in the submodels, and finally, using the above-mentioned submodels, a complex system management model could be constructed.

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USE OF CHLOROPHYLL/SECCHI DISC RELATIONSHIPS

M.W. Lorenzen

A number of researchers have proposed the use of empirical relationships between Secchi Disc and chlorophyll-a to project transparency changes to be expected from changed chlorophyll levels. There are data available relating Secchi Disc transparency to chlorophyll concentrations, see Oglesby and Schaffner (1975). The authors correctly noted that, for high concentrations of phytoplankton, reductions in their concentration produce little change in transparency. However, the implication that, for low phytoplankton concentrations, changes in their concentration would produce large changes in transparency is not necessarily true.

More recently, Carlson (1977) presented similar data, and developed the following regression equation:

$$\ln(\text{Secchi Disc}) = 2.04 - 0.68 \ln(\text{chl-a}) \quad . \qquad (1)$$

Carlson noted that the data did not fit an equation of the form

$$\frac{I_z}{I_o} = e^{-(\alpha + \beta C)z} \quad , \qquad (2)$$

Where I_o is the surface light; I_z is the light at depth, z; α is the extinction coefficient from factors other than algae; β is the incremental extinction coefficient from algae; and C is the algal concentration. It was assumed that I_z is equal to $(0.1)I_o$, when z is the Secchi Disc depth. He postulated that the inability to fit the data to this type of function may have been caused by the changing chlorophyll content per cell, according to the light conditions. It is possible that the value of α (nonchlorophyll light absorption) was quite different for the various lakes included in the data base.

The following theoretical analysis illustrates the importance of the extinction coefficient, α. It also shows that the large variation in Secchi Disc depth at low chlorophyll concentrations is likely to be a result of factors other than changes in chlorophyll concentration. Assuming that the Secchi Disc depth can be approximated by the depth of 20 percent surface light (Lorenzen, 1978), the Secchi Disc depth (SD) can be expressed as:

$$SD = \frac{-\ln(0.20)}{\alpha + \beta C} \quad (3)$$

At high algal concentrations ($\beta C >> \alpha$) the extinction of light and Secchi Disc depth are controlled by the phytoplankton. (It should be noted that although the Secchi Disc depth is inversely proportional to the phytoplankton concentration, the constant of proportionality is very small. For example, at a chlorophyll-a concentration of about 50 μg/liter, and with very clear water (α = 0.1), the change in Secchi Disc depth resulting from a 10 μg/liter increase in chlorophyll-a is only 0.15 meters [(1.61/1.61) - (1.61/1.90)]. For more turbid waters, the change would be even less.) However, at low algal concentrations, the penetration of light is largely controlled by light-absorbing properties other than phytoplankton, α.

The extinction coefficient due to factors other than algae, α, varies widely, ranging from 0.04 per meter for distilled water to 1.5 per meter or more (Clark, 1954, p.195). The incremental extinction coefficient, β, has been found to be fairly constant at approximately 0.2 mg/liter/m ash-free dry weight (Lorenzen, 1972). Unfortunately, the chlorophyll content of algal cells is quite variable, so that it is more difficult to relate the attenuation coefficient to chlorophyll. For illustrative purposes, Figure 1 was constructed for various values of α (0.04 to 1.0

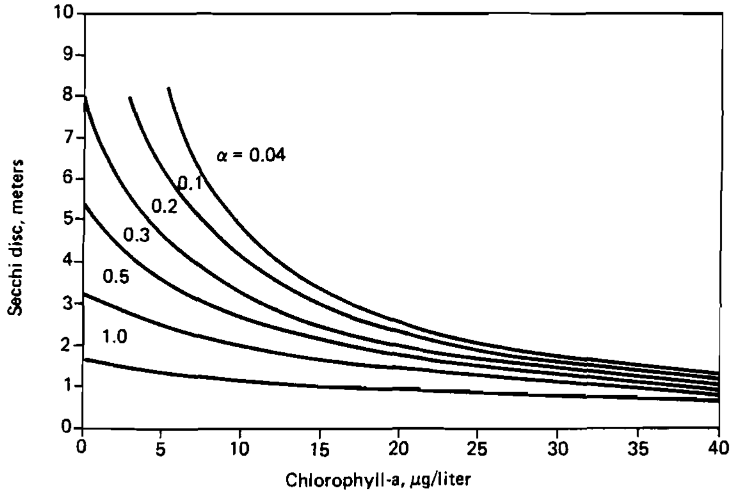

Figure 1. The theoretical relationship between Secchi Disc and chlorophyll.

per meter) and β represents 0.030 mg/liter/m chl-a (assuming that chlorophyll-a is equal to 1.5 percent of ash-free dry weight). For any particular lake, a reduction in algal biomass would be reflected by increased transparency according to the appropriate line for α. Dramatic increases in transparency at low chlorophyll levels would only be observed in waters very low in color or turbidity.

The analysis presented illustrates that the use of observed data for chlorophyll-a Secchi Disc obtained from different lakes can result in misleading projections. It is suggested that the value of α should be obtained for the lake being studied. An analysis (see Figure 1) can then be used to estimate the effects of changed chlorophyll on transparency.

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