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Eutrophication of Shallow Lakes: Modeling and Management. The Lake Balaton Case Study

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Proceedings of an International Workshop
L. Somlyódy, S. Herodek, and J. Fischer, *Editors*

EUTROPHICATION OF SHALLOW LAKES: MODELING AND MANAGEMENT

The Lake Balaton Case Study

Proceedings of a Workshop Organized by
the Hungarian Academy of Sciences, IIASA,
the Hungarian National Water Authority, and
the Hungarian Committee for Applied Systems Analysis,
29 August–3 September 1982, Veszprém, Hungary

Edited by L. Somlyódy
in collaboration with S. Herodek and J. Fischer

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PREFACE

Man-made eutrophication of lakes caused considerable water quality problems in many countries during the past 10-15 years. The International Institute for Applied Systems Analysis /IIASA/ initiated a program on studying the eutrophication of shallow lakes. The phenomenon of eutrophication is more irregular in character and less satisfactorily understood for shallow water bodies than that for deep lakes. The research was focussed on Lake Balaton as the subject of a case study. There were three main considerations which promoted the selection of Lake Balaton as case study:

/i/ fairly high amount of data was available already at the beginning, due to the preceding Hungarian research activities,

/ii/ the lake possesses the "typical" properties of shallow lakes in many respects,

/iii/ serious economic interests are associated to the solution of the eutrophication problem of Lake Balaton and so several "practical" questions had to be answered, in addition to the pure "scientific" issues.

The following institutions were involved in the cooperative research: IIASA, the Hungarian Academy of Sciences, the National Water Authority and the research institutes of those, primarily the Computer and Automation Institute, the Balaton Limnological Research Institute and the Research Centre for Water Resources Development. Thanks to the nature and structure of IIASA, as many as twenty foreign scientists e.g. from the USSR, USA, the Netherlands contributed to the research. From the Hungarian side nearly thirty scientists were invited for cooperation for longer or shorter periods of time.

The structure of the research was based on a systems analytical approach which allowed the joint study of scientific and practical issues covering a wide range of different disciplines such as biology, chemistry, physics, hydrology, mathematics, economics etc.

The mathematical models developed in close harmony and interaction with data collection and experimental work played a significant role in the project. Several existing methods were adopted for modeling the ecological and hydrophysical processes as well as water quality management in parallel with establishing new methodologies, too.

The cooperative research was completed at mid 1982. Round 70 scientists from 14 countries and 20 decision makers participated on the closing workshop held in Veszprém, Hungary. The results and findings - both "special" and "generalizable" - gained in the frame of the Lake Balaton study, were reported here and are summarized in the proceedings.

Scientists arrived at important conclusions for the water quality problem of Lake Balaton. These are relating among others to the external loading, the nutrient release of sediment /internal load/ and its "delaying effect", the influence of uncertainties of various origin and the establishment of the short-term control strategy complementing the existing protection measures.

The completion of the research was successfully scheduled. At the beginning of 1982 an expert committee was established in Hungary with the objective of elaborating recommendations for the government concerning the revision and modification of the existing measures on water quality control and regional development. The findings and conclusions of the Lake Balaton Case Study were especially of interest for the expert committee. This January the Council of Ministers approved the recommendations and made the relevant decisions. Thus, the research results were transferred to decision makers within a very short period of time allowing them to bring in scientifically well established new decisions.

The study on the eutrophication problem of Lake Balaton illustrated well, how effectively IIASA - an international scientific institute - can coordinate and perform a research having an impact on both science and application. I sincerely wish for IIASA to have many similar examples in its future history.

István Láng
Deputy Secretary General
of the Hungarian Academy of Sciences
Council Member of IIASA

CONTENTS

INTRODUCTION

Man-made eutrophication has been considered as one of the most serious global water quality problems of lakes during the past 10-20 years. Increasing discharges of domestic and industrial waste water, the intensive use of crop fertilizers and the rise in airborne pollution can be mentioned among the major causes of this undesirable phenomenon. The typical symptoms of eutrophication are among others sudden algal blooms, water coloration, floating water-plants and debris, excreation of toxic substances causing taste and odor problems in the production of drinking water, and fish kills. These symptoms can easily result in limitations of water use for domestic, agricultural, industrial or recreational purposes. Consequently, eutrophication may lead to a devaluation of the lake-shore region, therefore the solution of this environmental problem is of direct economic concern.

In 1977 and 1978 three state-of-the-art workshops were held at IIASA (Beck 1978, Jϕrgensen and Harleman 1978, and Jϕrgensen 1979)[1] which overviewed the water quality problems

[1] Beck, M.B. (1978) Mathematical Modeling of Water Quality. IIASA, Collaborative Paper, CP-78-10
Jϕrgensen, S.E. and D.R.F. Harleman (1978) Hydrophysical and Ecological Models of Deep Lakes and Reservoirs. IIASA, Collaborative Paper, CP-78-7
Jϕrgensen, S.E. (1979) Hydrophysical and Ecological Models in Shallow Lakes and Reservoirs. IIASA, Collaborative Paper, CP-78-14

of deep and shallow lakes. Based on the conclusions of these meetings a research on the eutrophication of shallow lakes was initiated at IIASA's Resources and Environment Area in 1978.

Deep lakes usually exhibit a remarkable year-to-year regularity in behaviour and the processes influencing this behaviour are quite well explored. Shallow lakes show much more irregularities which are attributed to the strong sediment-water interaction enhanced by wind, to the relatively deep photic zone within the total depth and in general to the significant sensitivity to the changes and fluctuations in environmental factors. In harmony with these features the eutrophication of shallow lakes is less satisfactorily understood than that of deep water bodies. The need for a comprehensive research on shallow lakes is further enhanced by the severe economic interest coupled with the region of those, especially in Europe, where the majority of lakes in the non-mountainous, density populated area are shallow and endangered by eutrophication.

Many nations are facing the problem of eutrophication, hence the exchange of experiences, methods and techniques beyond the national boundaries in an international setting such as provided by IIASA was expected extremely profitable. Furthermore, the application of systems analysis techniques is very promising in this type of problems for a variety of reasons. The complexity of the biological, chemical and hydrophysical processes, the strong interactions of the processes, the stochastic variability due to hydrometeorologic influences and the need for establishing feasible management strategies are just a few of the reasons why a systematic and comprehensive analysis is required. Another factor of interest is that lake-watershed systems are usually so large, that the possibilities for field experiments are often restricted by serious constraints. As a consequence, the information on the system should be retrieved from available data which are in general infrequent and error corrupted. Data analysis, including mathematical modeling techniques is therefore an important issue.Uncertainty should be faced and included in the study on the level of both understanding and decision making.

As the major case study of IIASA's activity, Lake Balaton, showing the unfavourable signs of artificial eutrophication, one of the largest shallow lakes of the globe was selected. This decision was stimulated by several reasons. Lake Balaton was thought a shallow lake typical in character in many respects which, in addition, provide a significant amount of data even at the beginning of the study. There has been a strong ongoing research and practical activity in Hungary. Again serious economic interests were associated to the solution of the eutrophication problem of Lake Balaton, the major recreational area in Hungary.

Thus, the problem of Lake Balaton raised both, scientific and practical questions and offered a possibility for drawing "general" conclusions.

From Hungarian side the Hungarian Academy of Sciences, the National Water Authority and the research institutes of those, first of all the Computer and Automation Institute, the Balaton Limnological Research Institute and the Research Centre for Water Resources Development participated in the cooperative study. The study was lead at IIASA by G. van Straten, Twente University of Technology Twente, the Netherlands in 1978-79, while by L. Somlyódy, Research Centre for Water Resources Development Budapest, Hungary between 1980 and 1982. The research was coordinated in Hungary by a subcommittee of the Hungarian Committee for Applied Systems Analysis. Apart from the permanent contribution of IIASA staff members, a number of collaborative links were established through IIASA with several external experts to contribute to the work and progress of the case study.

The results and progress of the study were presented and discussed on three Task Force Meetings in 1978, 1979 and 1981, resp. Altogether 21 reports were published /see Appendix III/.

Several meetings and discussions were held also with the representatives of the instutions and agencies responsible for Lake Balaton and its region. The major "practical" results of the study were presented for the Hungarian top level decision makers on a Colloquium at IIASA, May 1982.

The closing workshop of the four years long research was held from 29 August for 3 September 1982 in Veszprém, Hungary. Findings for Lake Balaton and other systems, were presented (see the Agenda, Appendix I). In addition a state-of-the-art discussion on eutrophication modeling was organized.

The workshop was followed and completed by a meeting of scientists and Hungarian decision makers responsible for Lake Balaton. The participants from 14 countries are listed in Appendix II.

The present proceedings is subdivided in two parts.

Part I summarizes the results gained for Lake Balaton through seven general reports. The first report by Somlyódy gives a background to and an overview on the work done. It also outlines the approach carried out for the study which is further detailed in subsequent general reports. These consider the problem of nutrient loads, sediment and its interaction with water, biochemical processes, hydrophysical processes, lake eutrophication models and finally eutrophication management models. Part II contains some selected papers presented on the workshop and the summary of the state-of-the-art discussion.

Part I

General Reports

MAJOR FEATURES OF THE LAKE BALATON EUTROPHICATION PROBLEM: APPROACH TO THE ANALYSIS

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1. INTRODUCTION

Over the last few decades Lake Balaton (Hungary), one of the largest shallow lakes in the world, showed the typical signs of artificial eutrophication. This is due to changes (e.g. urbanization, tourism and agricultural development) in the surrounding region and the increased nutrient loadings associated with them. The problem of Lake Balaton raised numerous scientific questions (the influence of shallowness, wind induced sediment-water interaction, spatial mass exchange, the combination of biological and hydrophysical phenomena in the modeling framework etc.). However, there is a strong economic interest vested in the lake and its immediate area (forming the largest recreational area of the country), which necessitates the solution of the real-life problem. Thus the study undertaken had to explore several "microscopic" scientific questions first in order to proper subsequent handling of management issues of "macroscopic" character. The study had to cover wide ranges of pro-

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* Research scholar at IIASA, 1980-1982

cesses and issues from algal dynamics to policy making.Thus the dilemma was how to incorporate all these issues in the analysis? In other words, which kind of approach should be adopted?

This introductory paper is divided into three parts. First the main features of the problem is discussed incorporating the watershed and the nutrient loads associated, the lake and the related characteristics and processes,spatial and temporal changes and finally data availability. This section is primarily borne on two summary reports in English (van Straten et al. 1979, van Straten and Somlyódy 1980) and original publications cited there. In the next section the systems approach based on the principle of decomposition and aggregation is introduced. This approach serves a basis for the case study. Subsequent General Reports discuss the major steps of this approach in a logical order from nutrient loads through biology to water quality management as illustrated in the last section outlining the entire effort.

2. CHARACTERIZATION OF THE LAKE-WATERSHED SYSTEM

Watershed and nutrient loads

The extension of the watershed area (incorporating also the lake) is 5776 km^2 (Fig. 1). Major subwatersheds, their areas and elevation differences are listed in Table 1. The

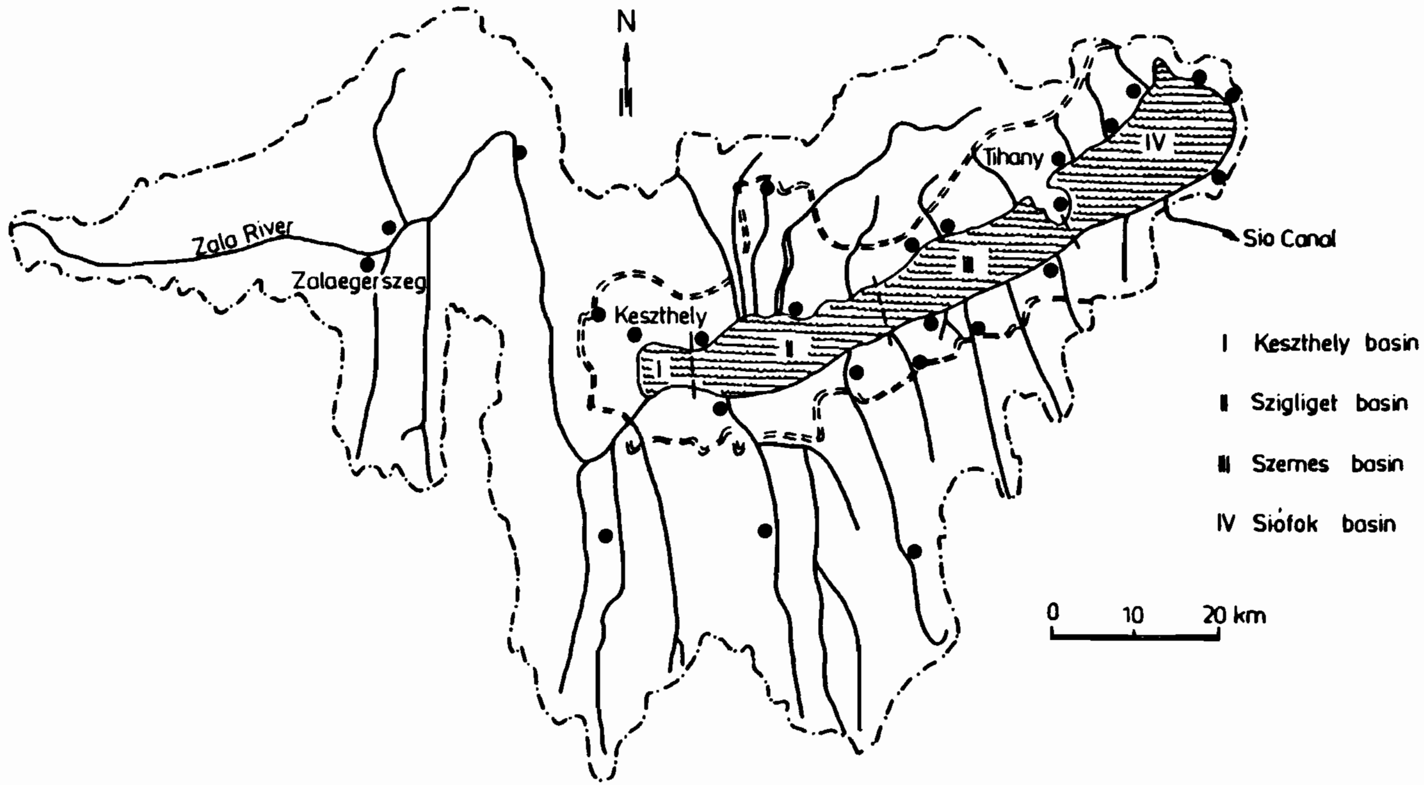

Fig.1 . The lake and its watershed

-.- boundary of the catchment
● significant sewage discharges
~~ tributaries
═ boundary of the recreational area

geological background is reflected in different ways in the subwatershed regions. The basin of the Zala River (Fig. 1) in the west mainly consists of hilly "Pannonian" alluvial land. The southern basin is dominated by low hills of sand and loess, but there are also marshlands. The northern basin is characterized by higher hills of limestone and dolomite. Here also the remainders of volcanic activity are found.

About the half of the land surface area is drained by the Zala River. Its average discharge, 9 m^3/s, represents 50 % of the total inflow for the lake. Tributaries of the Szigliget basin (Basin II, see Fig. 1) provide some 30 % of the inflow. The surface area of the four typical lake basins, A_{Li}, (Fig. 1) and that of the corresponding subwatersheds, A_{wi} show a change opposite in character from west to east (see Table 3). The ratios A_{wi}/A_{Li} are roughly 72:11:3:1 for Basin I ... IV already suggesting much larger relative nutrient load and less favourable water quality conditions for the western segments of the lake.

Erosion has a significant role in the changes of the lake and its basins. Due to the geological and topographical conditions, the soil loss is relatively high in the northern and moderate in the southern subwatershed. According to various estimates the annual soil loss ranges between 40-60 t/ha with lower and upper extremes of 25 and 170 t/ha, resp. The erosion rate depends on many factors and varies highly

	Area $[km^2]$	Elevation difference [m]
Zala watershed	2622	340
Northern streams watersheds	820	47-711
Southern streams watersheds	1176	117-212
Direct shoreline watersheds	562	-
The lake	596	-
Total catchment	5776	

Table 1. Subwatershed area and elevation difference (after Baranyi, 1975)

Land use type	Percentage of total catchment area	Percentage of slope categories				
		<5	5-12	12-17	17-25	>25
Plough land	35	68	20	8	3	1
Meadow	8	100	-	-	-	-
Pasture	7	73	12	5	7	3
Orchard and vineyard	6	41	28	17	11	3

Table 2. Slope distribution of various land use types (taken from Horváth and Kamarás, 1976)

from site to site. Apart from the natural factors such as slope, man influenced factors such as the type of vegetation or crop grown, and land use practice, have a significant effect on the erosion. Table 2 gives an impression of the various cultivation branches and its slope distribution.

Generally 1 to 3 % of the applied fertilizer is lost and reaches the lake through runoff and erosion. Thus the growing use of fertilizers increases the nutrient loads and is one of the causes of artificial eutrophication. Based on statistical records, around the mid 70th the fertilizer use for the total catchment was about 15 000 t P/yr showing a (6-7)- times growth compared to 1960. Taking the year 1950 as a reference basis, the increase is (60-70)-times.

Another spectacular change in agricultural production is resulted by the wide-scale introduction of industry-like livestock breeding. Thirty years ago large scale animal farms did not exist in the region at all; nowadays their number exceeds 40 with some total of 100 000 animals. From liquid manure generated here as much as 1000-1500 kg/d phosphorus loads the environment, 5-20 % of which may then reach the lake.

The other important factor characterizing the region of Lake Balaton is tourism. It is concentrated on the recreational area indicated on Fig. 1. The permanent population is 405 000, of which some 120 000 people are associated to the direct surrounding of the lake. The number of visitor days was about 8 million in 1978 which doubles the population and also the sewage discharge) during the relatively short summer peak season, July and August. Tourism increased by a factor of 14 between 1950 and 1978. The drinking water supply shows a five times development compared to 1960, resulting that the demands are nearly completely satisfied in the recreational area. The capacity of sewerage and waste water treatment increased in the past, too, still it has fallen behind the development of drinking water supply. All these factors resulted in a growing amount of nutrients reaching the lake. Although direct observations are not available, still it can be estimated that the load of the water body increased by an order of magnitude during the past 20-25 years. This change induced then the man--made eutrophication of Lake Balaton.

The nutrient loads and their distribution among the basins of the lake is difficult to quantify. When performing a source evaluation, all the emissions (fertilizer use, liquid manure, sewage discharges etc.) and transmission processes in the watershed should be considered which contribute to the external nutrient load of the lake. Another approach can be based on evaluating the data of the monitoring network of loads (tributaries, sewage discharges, atmospheric pollution etc.) entering directly the lake. This latter method is more straightforward but does not provide information on the origin of nutrients in question.

The uncertainties of the procedure mentioned secondly derive from infrequent observations. In general, one sample is taken monthly in the 20 major tributaries illustrated on Fig.1. For sewage discharges (see Fig. 1. for the location of more significant sources) even less observations are available. Accordingly the data set does not reflect the influence of floods or peak waste water releases although as stated by Joó (1976) and Jolánkai (1976), floods account for 60-70 % of the annual phosphorus and nitrogen loads. Based on the regular (scarcy) observations Jolánkai (1976) derived 260 and 910 t/yr loads for total phosphorus (TP) and nitrogen (TN) resp., while Joó (1976) taking also into account the information from the daily observations of the Zala River initiated by the West Transdanubian District Water Authority (WTDWA) extrapolated 580 and 2900 t/yr for the entire lake. The estimation based on source evaluation resulted in similar ranges (van Straten et al. 1979), thus we can conclude that the yearly average TP and TN loads lie between 260-580 t/yr and 910-2900 t/yr, resp.

A considerable portion of nutrients (e.g. 85 t/yr for TP in an average for 1976-81 based on the observations of WTDWA) reaches the lake through the Zala River and loads the Keszthely bay (Basin I, Fig. 1) the volume of which is ten times smaller than that of Basin IV. Consequently, the volume related loads of basins decreases from west to east inducing a typical longitudinal gradient in many water quality components (see later).

The amount of nutrients transported by rivers to the lake changes considerably year by year depending on the hydrologic regime. The yearly average TP load by the Zala River ranged between 55 and 114 t/yr during 1976-81 (trend-like changes are hard to determine from the existing records).

Geometry of the lake. Water temperature

The main geometrical parameters of the lake are: largest length 77.8 km, average width 7.7 km, surface area 596 km^2, average depth 3.1 m and volume 1861.10^6 m^3. It is a typical shallow lake of elongated shape. The water depth is smaller than 1.5 m at 9 % of the surface area, and does not exceed 3 m at 30 % of that. The largest depth, 11.6 m, can be found at the Tihany peninsula.

From hydrological and water quality aspects four basins of different character can be distinguished (Baranyi 1974): the Keszthely-, Szigliget-, Szemes- and Siófok basins Fig. 1, I ... IV . The geometric data of each basin are summarized in Table 3. This segmentation is often used later for the purposes of data collection and the development of various mathematical models.

Due to the small depth large annual fluctuations can be observed in water temperature. In an average a two-month long ice covered period exists. During summer, however, the temperature may exceed 25 ^{o}C. Due to the strong wind action thermal stratification practically does not occur.

Basin of the lake	Volume $[10^6 m^3]$	Depth [m]	Surface area of basins A_L $[km^2]$	Surface area of corresponding subwatersheds A_W $[km^2]$
I	82 (4.3 %)	2.3	38 (6.4 %)	2750 (53.1 %)
II	413 (21.8 %)	2.9	144 (24.4 %)	1647 (31.8 %)
III	600 (31.6 %)	3.2	186 (31.1 %)	534 (10.3 %)
IV	802 (42.3 %)	3.7	228 (38.1 %)	249 (4.8 %)
Lake Balaton total	1907 (100 %)	3.2	596 (100 %)	5180 (100 %)

Table 3. Geometric data of the four Lake Balaton basins at average level (104.9 m.a.s.l.) and the corresponding watershed surface areas

Hydrology

On a yearly basis the total inflow (957 mm) balances the evaporation losses (916 mm). The lake has one single outflow at the Sió control gate (Fig. 1). The outflow (671 mm) is about equal to the precipitation (628 mm).

The exchange time of water (Baranyi 1979) increases from west to east. It is 1 year for the Keszthely bay, 9 year for the Siófok basin, while 4,5 year in a lake wide average. The filling time (volume/inflow) often used in water quality management practices is considerably smaller than the exchange time; 0.25, 0.72, 0.97 and 1.31 year for the four subsequent basins (about 2 years[1] for the entire lake).

Hydrodynamics

Water motion in the lake is characterized by the slow hydrologic throughflow (~0.1 cm/s) and the pronounced wind induced circulation (the velocity ranges between 5-20 cm/s, with an extreme of 1 m/s at the Tihany Straight). The governing wind direction is N-NW. The monthly mean of the wind speed is about 2-5 m/s, but during storms it can exceed 25-30 m/s. The number of stormy days in a year when the velocity is larger than 8 m/s is about 60. The wind field is characterized by significant temporal and spatial non-uniformities (Béll and

[1] Note that the sum of inflows for the four basins is larger than the inflow of the entire lake. This is the reason why the filling time of the lake exceeds that of the individual basins.

Takács 1974). The latter is a consequence of the sheltering and deviating effects of the northern hills. The water practically cannot be found in stillstand. According to Muszkalay (1979) the number of longitudinal- and transversal seiches in a year reaches 1000. For longitudinal wind conditions, the largest difference in water level observed along the lake is around 1 m,an extremely high value. During such events the volume of Basin IV can change by 10 %, for a short period of time.

Due to geometric features of the lake and non-uniformities of the wind field a circulation pattern complicated in space is formed. It is characterized by large scale eddies and gyres - as shown by physical model experiments (Győrke 1975) and satellite photographs - which seems rather to isolate the bays and basins of the lake than to strengthen mixing. No quantifiable results on spatial and temporal velocity changes are available for the whole lake. Again, the extent to which these changes influence water quality is unknown. From water quality observations it can solely be stated that the intensive wind induced water motion is still not strong enough to level out the typical longitudinal water quality gradient (see later).

Beside the components of water motion discussed briefly above shallow waves should also be mentioned. Their height may reach 1 m (Muszkalay 1973). This phenomenon can play a role in sediment resuspension.

Water quality

The chemical properties of water reflect the mineral composition of the watershed. The calcium and magnesium bicarbonate content is rather high. Consequently the m-alkalinity is also high (4-5 mval/l) and the lake is buffered effectively at pH 8.3 to 8.7. The biogenic lime precipitation influenced also by the activity of the ecosystem is considerable.

Similarly to the changes of Ca^{2+}, most of the water quality components show a decreasing concentration profile from Keszthely to Siófok. This gradient is illustrated for TP and Chlorophyll-a in Fig. 2 on the basis of the regular observations of VITUKI (Tóth 1974). The same tendency can be found for total dissolved P, particulate P, suspended solids concentration or Secchi disk depth (the latter ranged in the seventies between 0.2 and 1.0 in Basin I). The explanation for the gradient lies in the volume related loads of basins which decreases from west to east. For illustration the total phosphorus loads (see Jolánkai and Dávid in this proceedings) of the basins are also given.

The deterioration of the lake's water quality followed the unfavourable changes (from an environmental view) of the region with a lag of a couple of years. At the early sixties the water quality was approximately uniform in the entire water body. The primary production was estimated at about 0.25 g C/m^2d (Böszörményi et al. 1962). Ten years later primary production reaches 0.6 g C/m^2d at Tihany and a gradient can already be observed.

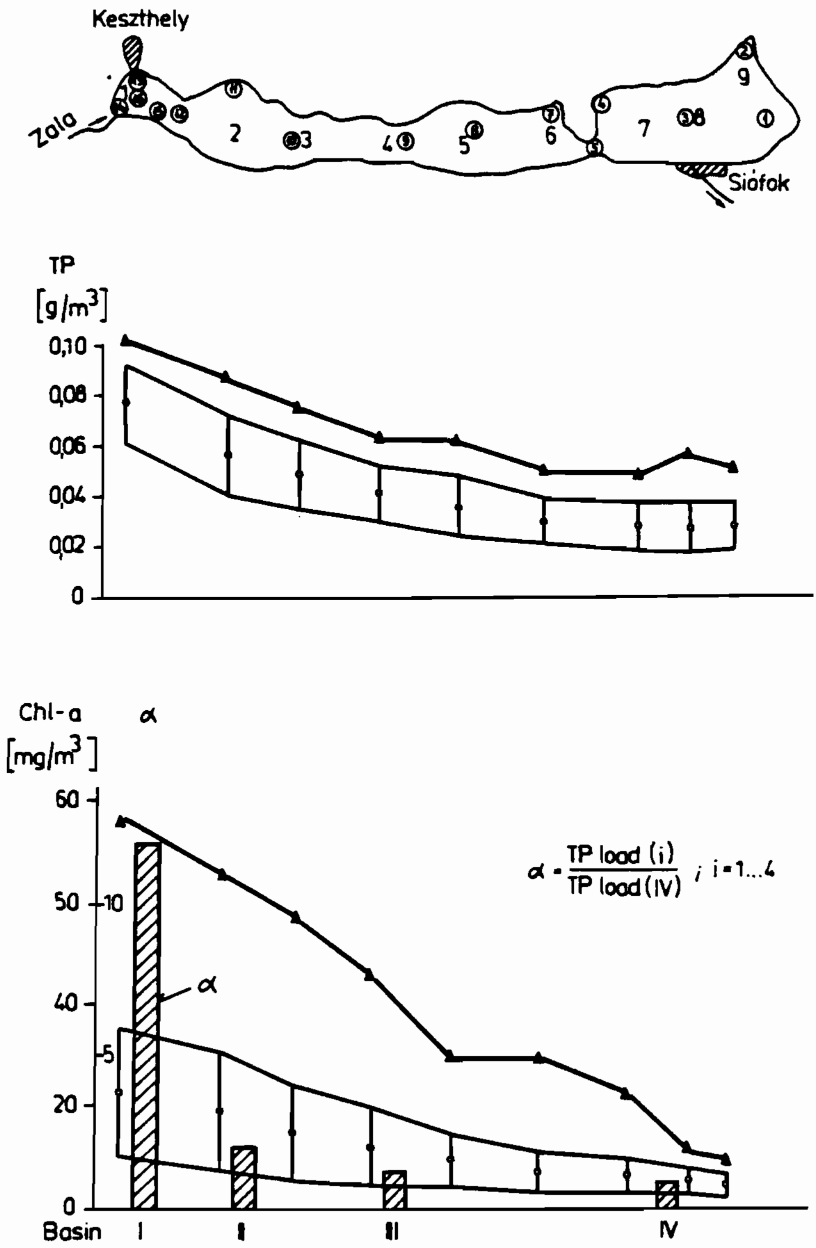

Fig. 2. Longitudinal distribution of water quality and phosphorus load (1976-1978). 1-9 and ③ sampling locations of VITUKI and Central Transdanubian District Water Authority, resp.

o mean, ⌽ ± standard deviation, ▲ maximum

The primary production in 1973, at Keszthely exceeds 13 gC/m^2d - a hypertrophic value, in 1974 at Szigliget 3 gC/m^2d, in 1976 at Szemes 2 gC/m^2d while in 1977 at Tihany 1.5 gC/m^2d (Herodek and Tamás 1976, 1978). Algal biomass is a subject of similar changes: in 1965 8 mg/l and 1 mg/l are observed at Keszthely and Tihany, resp. The corresponding values in 1977 are 60 mg/l and 7 mg/l (Vörös 1979). The chlorophyll data show the same trends (Felföldi 1981). As it is apparent from the data,the man--made eutrophication proceeds not only in time but also in space from west to east.

The changes of the last 13 years can be better traced based on regular measurements of VITUKI at nine sampling locations (see Fig. 2). Fig. 3. illustrates the changes in the annual mean and extremes of the Chlorophyll-a concentration for the four basins (sampling locations 1,2,5 and 8). The maximum values observed during the recreational period (May-September) are also given in the figure for Basins I ... III. From a linear trend analysis the annual deterioration of water quality in terms of the mean is 23 % in a lake wide average (it is 26 % for Basin I and 19 % for Basin IV). From this figure it appears that the mean Chlorophyll-a concentration characterizing properly the water quality of the lake increased by a factor of about 10 during the past 10 years; a striking value.

Fig. 3 incorporates also the trophic classes (in terms of the peak Chl-a concentration) proposed in the final report

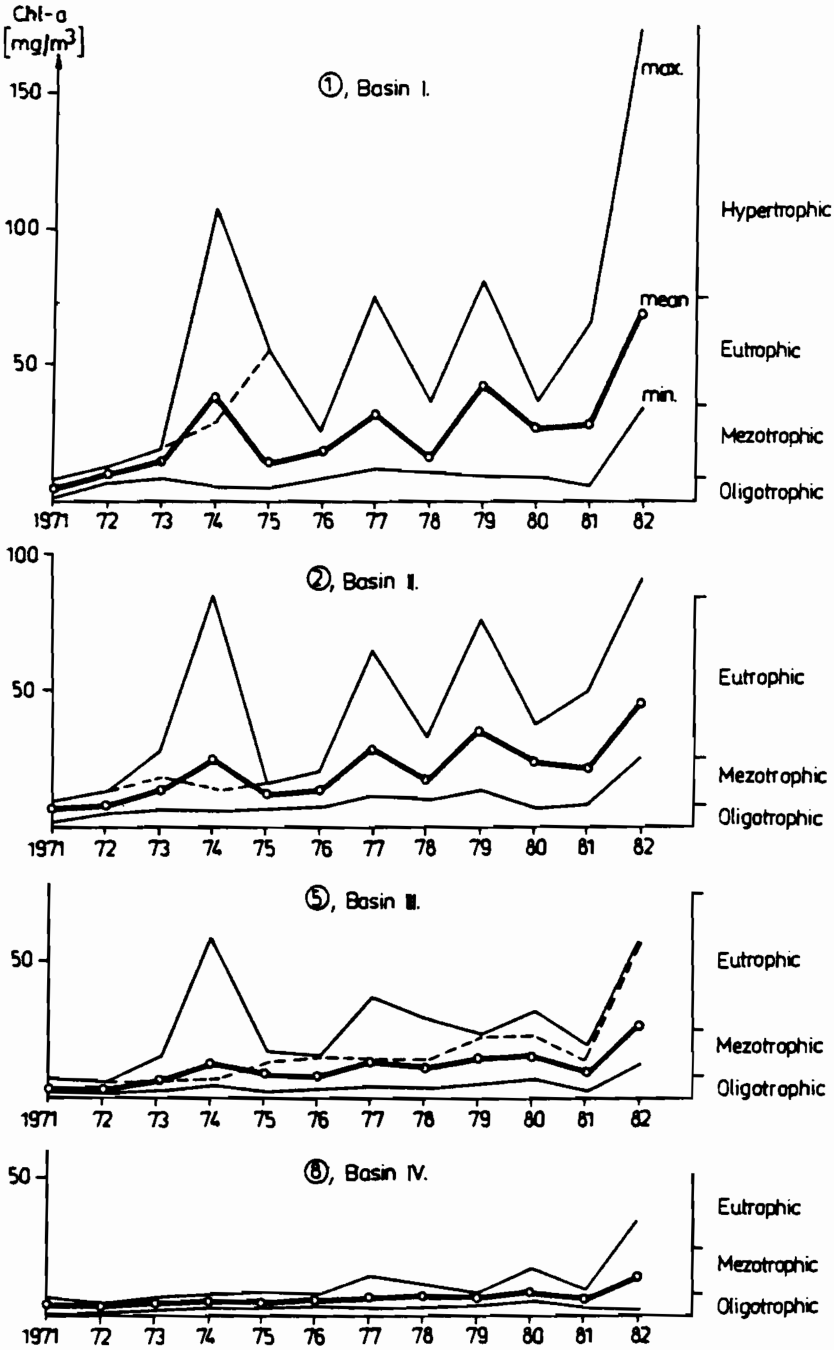

Fig. 3. Change of chlorophyll-a from 1971 (sampling locations 1,2,5 and 8, see Fig. 2)
--- maximum for May-September

of the OECD[1] (1982) eutrophication study. Accordingly the Keszthely basin became eutrophic in 1973-74, while hypertrophic during the last year. Basin II is eutrophic since 1976, nowadays hypertrophic, too. The water of Basin VI shifted to be eutrophic around the end of the seventies, while that of Basin IV in 1982.

Fig. 3. clearly indicates, how strongly the water quality of the lake is affected by meteorologic factors. The figure also reflects sampling problems. In an average the number of observations in a year is ten. Under such conditions the computation of the yearly mean is rather uncertain and even more so as the dynamic properties are considered. As seen from Fig. 3., the observation of an algal bloom in 1974 resulted in larger Chl-a mean value for the entire year than for the recreational period.

Fig. 4 compares the basins of Lake Balaton to other lakes. Following the structure of the empirical model of R.A.Vollenweider (OECD 1982) the maximum Chl-a concentration is plotted against the normalized annual total P load, P_{TL} (note that P_{TL} is computed from the actual total P load, the hydraulic load and the filling time; see e.g. van Straten in this volume). In Fig.4. one of the plots of the final report of the OECD eutrophication study (OECD 1982) served as a basis to which data pairs of Lake Balaton were added. For the computation the observations of 1975-79 were used and the extreme situation found in 1982 is also illustrated.

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[1] Organization for Economic Cooperation and Development

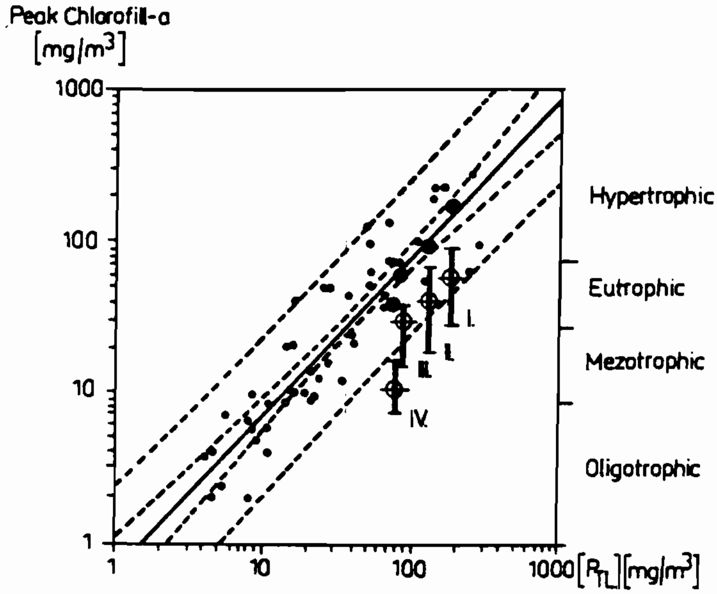

Fig. 4. Comparison of Lake Balaton to other lakes

I...IV	Basin I...IV
o	mean for 1975-79
I	domain defined by extreme values
●	peak concentrations in 1982
•	data pairs of OECD (1982) lakes

As seen Lake Balaton do not separate from the domain of OECD lakes in the double logarithmic scale. The significant fluctuation in water quality and the large peak values of 1982 are apparent although both are damped by the scale employed.

Phosphorus played the limiting role in the eutrophication process till the end of seventies. Nowadays nitrogen and light conditions are becoming more and more important in the limitation. This is an obvious consequence of the development of eutrophication (Oláh et al. 1981, OECD 1982).

Chlorophyll-a, biomass and primary production observations reflect a fast dynamics of biological processes. Generally two algal peaks can be found in a year, in spring and around the end of summer. In spring diatoms dominate while in summer a mixed population occurs (concerning the structure of algal population the reader is referred to Vörös (1979) and Vörös and Németh (1981)). At present, already blue-greens dominate this population. The seasonal change of algal biomass is similar from year to year, however the actual dynamics largely depends on meteorologic conditions causing significant deviations in the observed peak values of the subsequent years.

The nutrient cycle generally is influenced also by consumers and decomposers. For Lake Balaton, however, the importance of consumers is small: the transport in the food chain is negligible in this direction and most of the organic material is decomposed by bacteria (for details see Herodek in the same volume)

As it was mentioned previously phosphorus played the decisive role in reaching the present level of man-made eutrophication. Simultaneously, phosphorus is the only nutrient through which the process can be controlled effectively in practice. For these reasons the efforts were focussed on phosphorus during the present study.

Relatively high portion of TP, 30-35 %, is found in dissolved form (DP) in the lake water. As the dissolved reactive P (DRP) is low all the time (2-10 mg/m^3; most probably because of sorption processes), a significant portion of DP is composed of organic P and/or condensated polyphosphates. The temporal changes of TP reflect the strong influence of wind induced resuspension affecting particulate phosphorus content.

For this reason neither TP nor - due to its permanently low level - DRP can be used for characterizing the eutrophication of Lake Balaton.

Sediment and its interaction with water

Round 90-95 % of phosphorus entering the lake is accumulated in the top layer of sediment. A considerable portion of this amount is not readily available for algae but with the progress of eutrophication it can be transformed to available forms. This makes the sediment of paramount importance in eutrophication control.

Lake Balaton sediment is composed mainly of $CaCO_3$ and fine sand. The organic material content is low, about 2 %, although roughly 1/3 of bacterial decomposition takes place in the sediment. The mean particle size of sediment material is about 20-30 μm (larger along the southern shore).

Water content of the sediment is high, 70-80 %. The concentration of total phosphorus is about 200-600 mgP/g dry material, while that of the total nitrogen is larger by an order of magnitude (Tóth 1976). TP decreases from north to south and the change in TN is opposite in character. The TP concentration does not vary significantly along a vertical in the sediment (Tóth 1978). Information on the changes of ecologically really important nutrient fractions are not available.

Due to the shallowness of the lake and the wind action, oxygen conditions are favourable. Generally aerobic conditions can be observed even in the top layer of sediment. Phosphorus is bound mainly to $CaCO_3$ and iron. Wind induced resuspension and diffusion play important roles in the sediment-water interaction. Resuspension depends on the turbulent kinetic energy available at the bottom and wave motion. A measure of resuspension is the fluctuation of the suspended solids concentration which can increase from the background value of about 5-10 mg/m^3 up to 100-200 mg/m^3 in the open water and 500 mg/m^3 in the off--shore region (Hamvas 1967, Győrke 1978). Coupled to this process also the nutrient release from sediment and light conditions change.

In summary, the role of sediment in eutrophication is not satisfactorily explored. This can be attributed to complicated biological, chemical and physical processes and to the difficulties in sampling and analytical techniques, as well.

Data basis and data bank

The characterization of the lake-watershed system given in the previous sections approximately reflects the data availability at the beginning of the Case Study. Unfortunately very few data were accessible on computer. The precondition for the present research (and perhaps also the first result to be mentioned) was the preparation of the data bank which was then gradually extended. In its present stage it incorporates among others - mainly for the period of 1971-1979 - water quality data of the networks of VITUKI and Central Transdanubian District Water Authority (9 and 16 locations, resp., see Fig. 1) ; water balance data (monthly means); meteorologic observations (daily temperature and global radiation, three-hourly wind data);daily nutrient load observations of the WTDWA for two cross-sections of the Zala River(completed by weekly data of more than twenty chemical components); load data for other tributaries and waste water discharges and the results of special observations performed in the frame of the study.

A preliminary analysis on the data revealed several difficulties already at the beginning. In many cases the temporal frequency and spatial density of observations is rather small.

The data often reflect certain local influences or the consequences of inappropriate averaging. Harmonized observations for physical, chemical, biological and hydrometeorological parameters - which would be of great importance from the viewpoint of understanding eutrophication - were not available. Thus, the significant role of data uncertainty had to be recognized in the modeling effort.

3. APPROACH TO THE STUDY

Based on the previous sections the water quality problem can be characterized by:

- complexity,
- interdisciplinary character,
- a variety of uncertainties,
- need for scientific understanding of various interrelated processes of quite different scales (e.g. the time scale ranges from several hours to decades)
- need for knowledges which can be utilized on the level of policy making.

Clearly, the approach developed for the research should reflect all these features.

As mentioned in the Introduction a systems analytical approach was employed with an intensive use of mathematical models. If solely the present state of water quality modeling is considered many difficulties , dark spots and gaps are encountered. Most of the models developed until now in the research context did not aim to provide solution for management questions. Complementarily, management models are often based on "a priori" assumptions without scientific justifications. Nevertheless, even if research models are considered several problems emerge (for details see Somlyódy 1982a).For example, how to combine properly the description of hydrophysical and biological processes in the model? What is the necessary and realizable detailedness along the spatial, temporal and ecological axes of the model taking into account available theoretical knowledges, observations and modeling techniques? How to handle the uncertainties and so forth?

In the frame of the present study all these problems had to be faced. Eventually an approach was established which is based on the principle of decomposition and aggregation (Somlyódy 1982b, 1982c). The procedure starts with a reasonable decomposition of the complex system into smaller, more tractable units which are accessible for separate and detailed studies (laboratory and in-situ observations, mathematical models etc). These units or issues form a hierarchical structure. The studies on these units are followed by aggregation, the aim of which is to preserve and integrate only the essentials for the subsequent levels of research based on the relative importance of subprocesses considered, ruling out the unnecessary details.

For the purpose of illustration water motion and its influence on water quality is mentioned. When analyzing water circulation the fine structural changes are traced having typical scales of some minutes or hours in time and say 1-1000 m in space. From the viewpoint of spatial and temporal changes of the water quality, however, it is often satisfactory to divide the lake into larger segments or basins and characterizes water motion through lumped convection and dispersion terms which are in harmony with the segmentation adopted and based on studies with detailed hydrodynamic model (see in this respect Harleman and Shanahan in this proceedings). Finally it can easily occur that even some simpler description of water motion is satisfactory for water quality management (e.g. hydrologic flow governs the long-term changes of water quality).

As a result of the decomposition-aggregation principle one can avoid the use of a "large" model difficult to handle, and a sequence of corresponding detailed and aggregated models can be applied instead. Only the aggregated models are coupled in an on-line fashion (the approach is off-line in character for the detailed models) thus possibly resulting in a relatively simple model at the top of model hierarchy where management issues are handled.

The application of the method for the Lake Balaton problem is explained through Fig. 5 (Somlyódy 1982b). The first decomposition that directly comes to mind is the distinction between

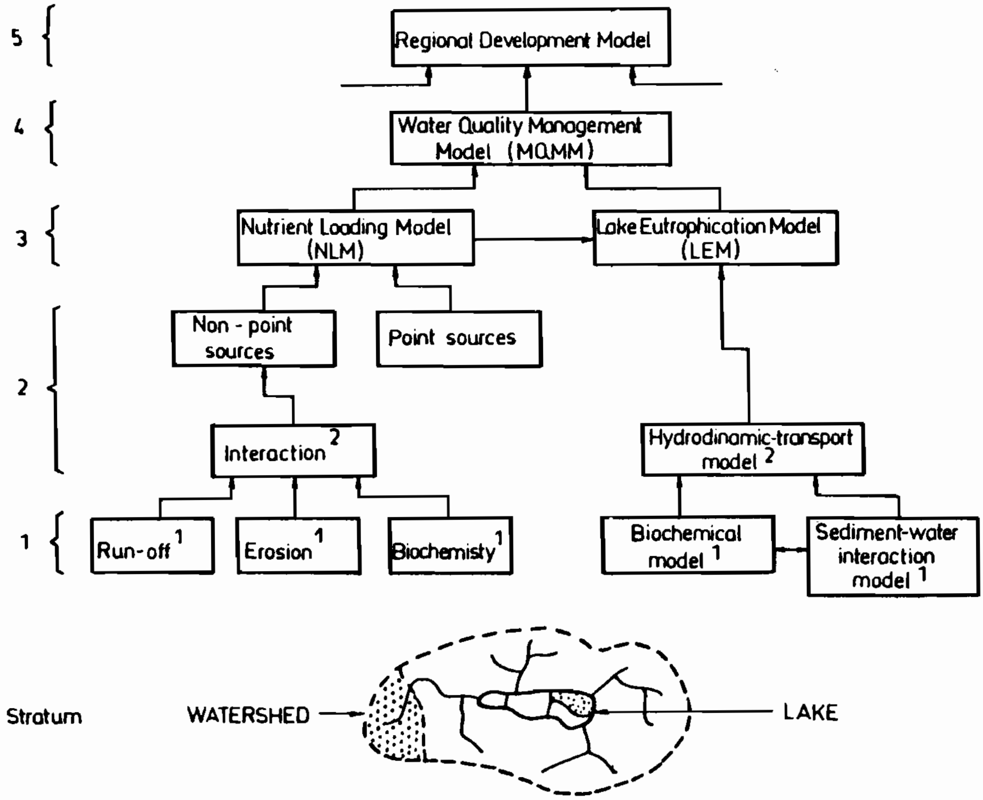

Fig. 5. Approach to the study: the principle of decomposition and aggregation

1 submodels for uniform segments (dotted areas),
2 coupling of the submodels

lake and watershed, since the water quality problem arises in the lake, but the causes and almost all the control possibilities come from and go for the watershed. Next, such segments are separated in the lake (and in the catchment) which are more or less homogeneous. As a following stage of analysis the biological and chemical processes, as well as those associated to the sediment of each segment are identified (the procedure is the same for the watershcd, see Fig. 5). In order to couple the uniform areas, interactions, i.e. in-, and outflows for the water body should be accounted for. These can be established on the basis of hydrodynamic and transport models. Therefrom the lake eutrophication model (LEM) and nutrient loading model (NLM) are resulted in on Stratum 3 of Fig. 5. Forcing functions are among others water temperature, global radiation, duration of the photoperiod, wind velocity and direction, precipitation and other parameters of water balance.

In the course of model development historical data are used for forcing functions. Of these the nutrient loads are the most important for the lake model. These represent the factor through which water quality can be controlled in practice. In the next stage of the analysis, when applying the model to answer real--life questions (e.g. evaluation of water quality under changed load conditions) future scenarios should be employed. At this step forcing functions reflecting some critical conditions (from the viewpoint of water quality) should be selected or they have to be considered as stochastic variables.

On Stratum 3 of Fig. 5 the dynamic response of lake segments is derived on various nutrient load functions. On Stratum 4 the water quality management model (WQMM) can be found which comprises somehow LEM and NLM. The objective of WQMM is to generate alternative management options and strategies (NLM is used here in a planning mode, i.e. control variables should be introduced) and to select among these alternatives on the basis of objective(s) under the given constraints. Both water quality and costs can be used as objective functions or constraints, and quite often their weighting is required.

One faces the formulation of an optimization problem on this level of the analysis the solution of which necessitates the aggregation of complex dynamic models on Stratum 3. The procedure employed for Lake Balaton is shown in the last General Report.

For the sake of completeness it should be mentioned that WQMM could be thought of as a part of a regional development policy model forming the top of the pyramid (see Fig. 5), a field which is beyond the scope of this study.

4. A BRIEF OVERVIEW ON THE RESEARCH ACCOMPLISHED

Various approaches and models can be applied on each stratum of Fig. 5. The development and intercomparison of alternative methods are very fruitful.

In the frame of the study three ecological models were developed using the observations of 1976-79. Models of Leonov and van Straten describe phosphorus cycle in water and sediment (see van Straten in this volume). The third model (Kutas and Herodek in this proceedings) is a mixed algal biomass--nutrient model incorporating nitrogen, too. Three further models were adopted and intercompared (together with van Straten's model) for the Keszthely Bay by Baker and Harleman (see this issue).

Models differ in complexity, their ecological concepts, number of state variables and parameters, and in the parameter estimation technique employed. For model development the research done at the Biological Research Institute (see Herodek in this volume, Herodek 1977, Herodek and Tamás 1978) and the results of regular observations were utilized as biology is concerned, while on sediment and its interaction with water the work performed at VITUKI during the past three years served as a basis (Gelencsér et al. 1983 and Lijklema et al. in this proceedings).

Lake eutrophication models distinguished four basins coupled by hydrologic throughflow and a lumped, wind dependent dispersion coefficient. The validity of these assumptions were checked by a coupled one-dimensional hydrodynamic-water quality model (see Harleman and Shanahan in this volume). Different three-, two- and one-dimensional hydrodynamic models were developed, of which the two-dimensional version incorporating depth integrated values were used after aggregation for the purpose of coupling.

Comprehensive data collection and field studies were performed in relation to the nutrient loads. From these information the load estimate of the lake and its basins were arrived at (Jolánkai and Somlyódy 1981 and Jolánkai and Dávid in this issue) which served as a unified bases for both ecological and management modeling.

In studying nutrient loads the daily observations of Joó (1980) on the Zala River played a central role.This data set allowed us to perform a time series analysis including precipitation, streamflow, suspended solids, total nitrogen and total phosphorus (Beck 1982), furthermore studying the uncertainty related to infrequent observations and the stochastic features of the load relying on the hydrologic regime (Somlyódy 1983).

From practical point of view water quality management is the most important issue. Five models were developed on this field (see Somlyódy in this proceedings).

Dávid and Telegdi (1982) use historical data for searching direct correlation between water quality and watershed development factors influencing nutrient loads.The model is a useful tool for analyzing the future development of watershed and the corresponding changes of water quality. Bogárdi et al. (1982) employed multiobjective programing for an agricultural subwatershed. The objective is to minimize phosphorus load as a stochastic variable on the one hand, and costs and economic losses on the other. Kovács et al. (see this issue) employed mixed integer programing for the design of sewer system of the recreational area. Hughes (1982) analyses the spatial distribution of investments (sewage treatment and reservoirs) using integer programing. Finally the author of this paper works out a method which is a result of the step by step application of the decomposition-aggregation principle. The model incorporates an aggregated stochastic load vs. water quality response model and its goal is to establish the short term "optimal" water quality control strategy of primary importance at the present state of water quality. For details the reader is referred to the last General Report of this proceedings.

SUMMARY

Over the last few decades Lake Balaton Hungary , one of the largest shallow lakes in the world, showed the typical signs of artificial eutrophication. The problem of Lake Balaton raised numerous scientific questions. However, there is also a strong economic interest associated to the lake and its immediate area, which requires the solution of the real-life problem. Thus, the approach adopted for research had to allow for detailed scientific analysis as well as analysis of management issues. The paper introduces the problem,describes the systems approach based on the principle of decomposition and aggregation; and outlines the research accomplished which is discussed in details in subsequent six General Reports of this proceedings.

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NUTRIENT LOADS AND WATERSHED DEVELOPMENT

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I. INTRODUCTION

To make any decision about the selection of water quality control strategies, and thus in our case eutrophication control strategies, one has to know what are the causes of the preceived unfavourable processes, what are the sources of this pollution and in what quantities are these pollutant discharged into the recipient water body in respect to both time and space. More precisely one could sketch the main interrelationships of the management process as shown in Figure 1. where the decisive and governing role of pollution load determination process is clearly indicated.

Accepting the generally known fact that external plant nutrient discharges are the causes of eutrophication then our task can be confined to the identification of nutrient sources and to the determination of loads attributable to these sources. On the watershed of a lake the sources of plant nutrients can be categorized as direct-indirect; and point-non-point sources and the combinations of these. This categorization marks also the choice of control options.

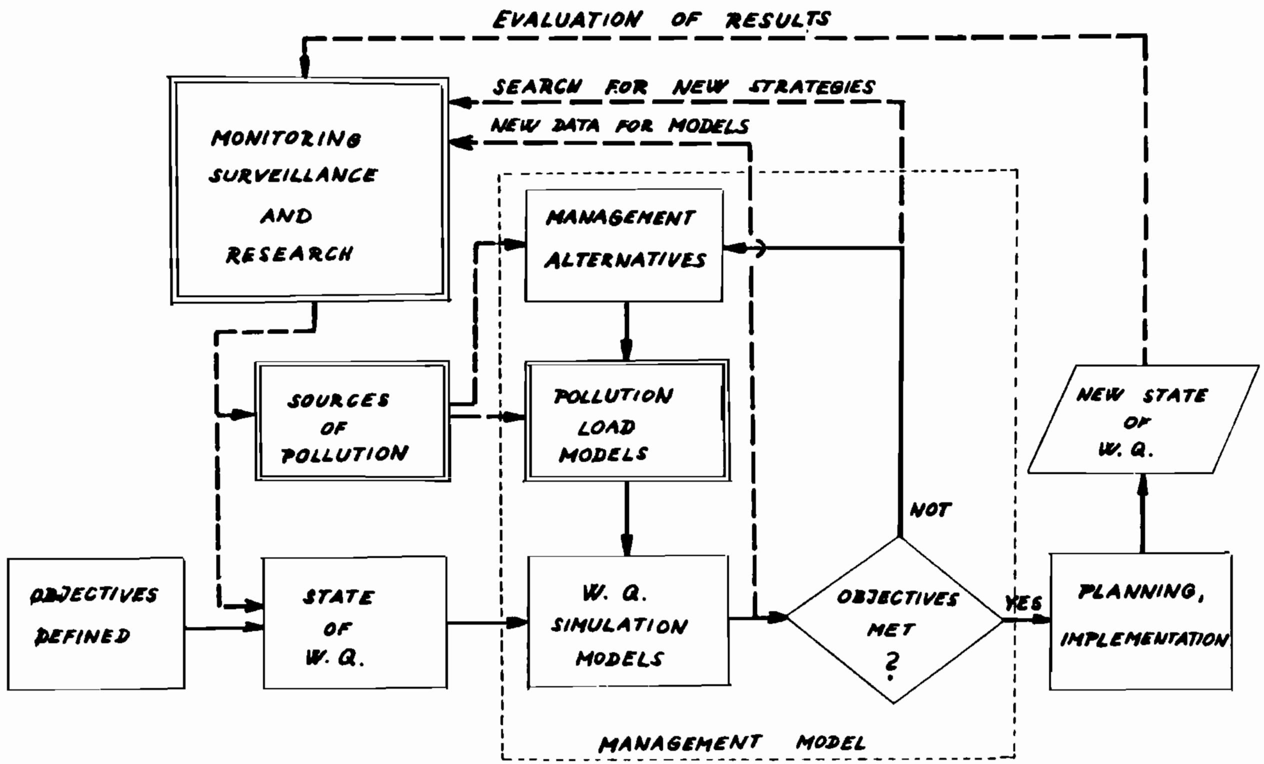


FIGURE 1. THE ROLE OF POLLUTION LOAD DETERMINATION IN THE OVERALL PROCESS OF WATER QUALITY MANAGEMENT

The objective of this general report is to provide a review of the work and studies carried out in the framework of the Balaton Case Study and to draw generalizable conclusion (if any) relevant to this specific field of, nutrient load determination and modelling. In doing so to major target of the work done will be emphasized, that is

i) to provide nutrient load data and relevant informations that will be the basis of eutrophication modelling and the decision making processes of eutrophication control and

ii) to achieve better understanding of the complexity of processes that have effects on nutrient loading rates originating from large drainage basins.

II. ESTIMATION OF NUTRIENT LOADS FOR THE LAKE BALATON CASE STUDY

The accuracy of the quantitative determination of pollution loads depends on the data base available and also on the technical/economical and time constrains imposed on carrying out further objective oriented field measurements that is on the monitoring and surveillance system. Further pilot scale studies are needed to reveal and quantify yet unknown but suspected sources and processes of pollution such as load dynamics or short and long-term time variability. The chances to apply any theoretical approach to the load determination are strictly defined by the data base and supplementary research possiblities.

In this section:

The data base available for Lake Balaton will be briefly described; next; long term average phosphorus loading figures will be presented as calculated on the basis of raw monitoring data and;

Methods used for correcting the basic loading figures to account for various uncertainties resulting from unfrequent observations and also from variations within the hydrological regime, will be briefly described.

2.1. The Lake Balaton data base for nutrient load determination

a) Nutrients carried by tributaries

The water quality monitoring network system involves 20 of the major tributary water courses entering the lake (Figure 2). With a catchment area of 4522 km^2 it comprises 87 % of the total watershed. An additional temporary sampling program covering all of the permanent water courses 32 was launched in 1982. The regular sampling usually occurs once a month when among others streamflow Q and all P and N forms are measured. In addition to streamflow measurements carried out simultaneously with the sampling more detailed information based on

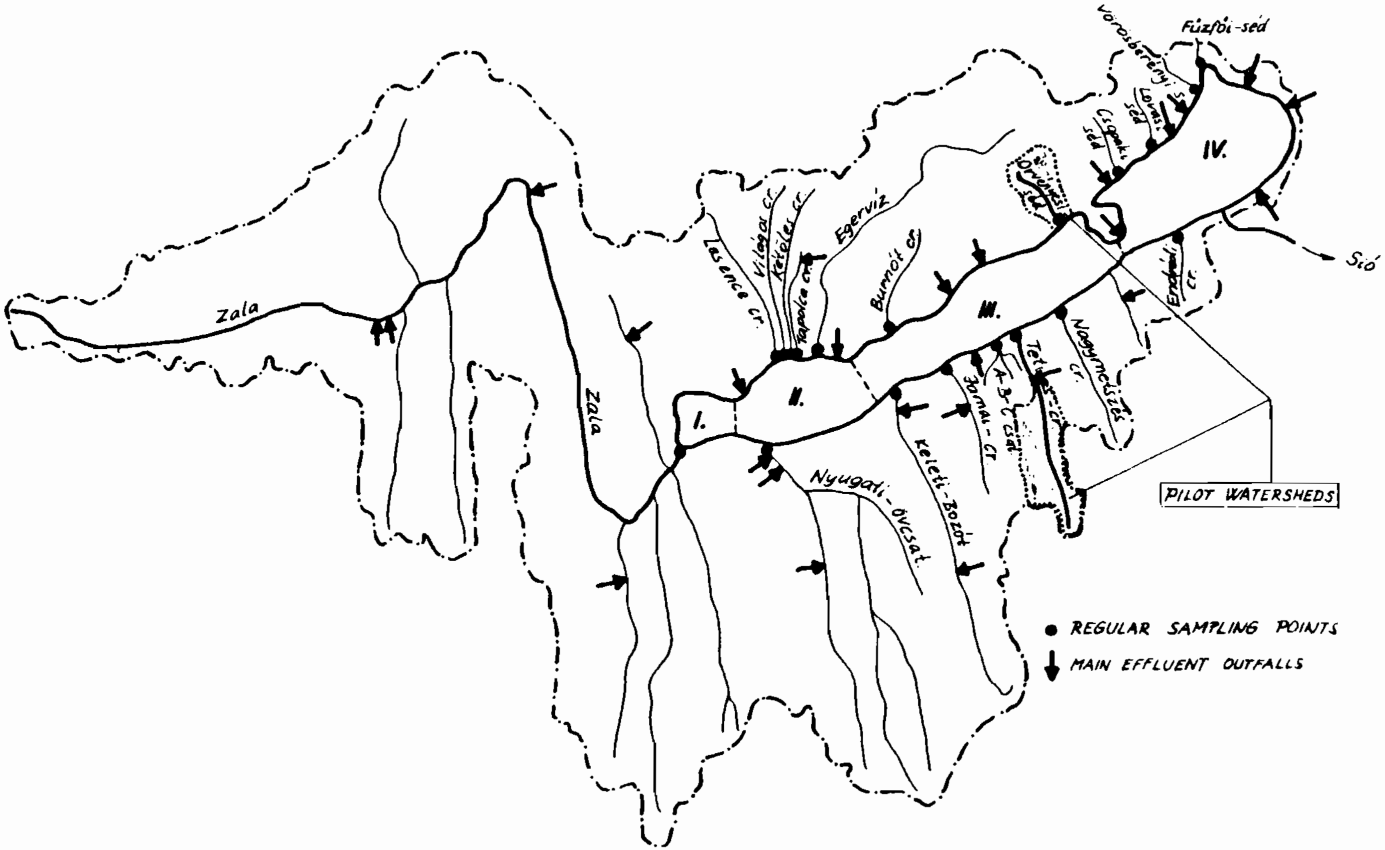


FIGURE 2. LAKE BALATON AND ITS WATERSHED

historical flow records were available for the 10 largest tributaries. Since 1976 exceptional data base was made available for the 10 largest tributary, River Zala to which about 50 % of the total watershed belongs. This data base included daily flow, TP, TN and TSS measurements and weekly complete chemical analysis (covering more than 20 components).

Since earlier studies (Joó, 1976, 1979; Jolánkai 1976, 1979) have shown that on smaller tributaries unobserved rapid flood-waves contribute a large part of the total nutrient load, field studies aimed on the quantification of these effects have been carried out on two pilot watersheds namely that of Cr. Tetves and Cr. Örvényes (Shaded areas in Figure 2.)

b) Sewage loads

The data of annual 2-6 measurements are available for the 46 largest effluent discharges on the watershed. These infrequent data provide basis only for a rough estimation. In addition to this, high variations in sewage load are appearent due to the tourism, and summer population along the recreational region of the lake. Summer population in the recreational region may be six-seven times higher than the permanent population. Detailed measurements on some of the largest sewage treatment plants along the shoreline gave account of this variability.

A special problem of sewage load determination is created by the intermittent release of nutrient rich waters from fish ponds and marsh lands receiving treated sewage (Keszthely, Fonyód, Lelle, STPs).

c) Direct non-point sources

Direct diffuse sources of nutrients may be very significant. Due to their specific nature and unidentifiable character some of load components can be only roughly estimated. Such load components are:

i) load carried by the stormwater runoff from residential,recreational urban catchment. Data of pilot scale studies are available (Botond, 1981, 1982)

ii) Surface runoff from the direct Balaton catchment (an area of about 570 sqkm, which is not covered, by the regular observation network of tributaries).

iii) Direct atmospheric input by wet and dry deposition. Load rates are estimated on the basis of measurement data of seven stations (Mészáros et al, 1980).

iv) Load represented by subsurface flow and infiltration from the direct shoreline.

2.2. The phosphorus loading figures

In accordance to those stated in relevant studies (Herodek 1977 and Somlyódy 1982) phosphorus is the most important nutrient in eutrophication and its control.Therefore on the basis of the above described data base monthly and annual P load values have been calculated. Due to the scarcity of data and to uncertainties of hydrological origin,long-term (1975-1981) average estimate values have been produced to meet the demands of ecological and management modelling, thus to provide basis for the selection of nutrient load reducing strategies.(Table 1.). In this table load values of total phosphorus and biologically available phosphorus (BAP), being the most significant component from the point of view of eutrophication are given. The definition of this latter is a subject of many disputes and for this project it has been given as follows

i) For sewage waters; BAP = Total phosphorus

ii) For streams; BAP = Orthophosphate phosphorus +20 per cent of the fraction between total phosphorus and PO_4-P.

Average loading figures of streams and sewage have been corrected to account for uncertainties araising from scarcity of data and also from hydrological variations. The basis of this correction will be briefly discussed in the following sub-section. For more details of this correction methods former works of Jolánkai (1975, 1977) Jolánkai and Somlyódy (1981) and Somlyódy (1982) should be consulted.

Based on the phosphorus loading values given in Table 1. and on the other related considerations, the following main conclusions can be drawn:

a) Long-term average external Tp and BAP loading rates are 865 and 465 kg/day respectively

Description		I.	II.	III.	IV.	Whole lake
		Basins of the lake				
Water courses	TP	230	153	62	10	455
	BAP	130	95	26	4	255
Direct sewage TP=BAP		3	6	8	78	95
Other sewage (through fish ponds and reservoirs) TP=BAP		-	-	25	-	25
Urban runoff	TP	12	35	39	74	160
	BAP	3	10	10	22	45
Other direct runoff (agricultural)	TP	28	32	40	26	126
	BAP	6	11	17	8	42
Atmospheric input	TP	3	12	16	19	50
	BAP	1	5	7	7	20
Total external load	TP	276	238	190	207	911
	BAP	143	127	93	119	487
Load per unit volume ($mg/m^3 day$)	TP	3.37	0.58	0.32	0.25	0.47
	BAP	1.82	0.31	0.16	0.15	0.26
Load per unit area ($mg/m^2 day$)	TP	7.26	1.66	1.02	0.91	1.54
	BAP	3.77	0.88	0.50	0.52	0.81

Remark: TP = Total Phosphorus; BAP = Biologically Available Phosphorus

Table 1. The phosphorus loading figure of Lake Balaton kg/day (average for 1975-1981)

b) The longitudinal distributin of external load is fairly uniform when the main basins of the Lake are considered as shown in Figure 2. A marked decrease of loading rates from the West to the East is appearent, however, if individual tributaries are considered.

c) From the last two rows of Table 1. showing volume and area specific loading rates respectively, steep drop of specific loading rates are apparent from Keszthely towards Siófok, showing close relationships with

  i) respective values of eutrophication measures and

  ii) with figures and indices expressing the level of watershed development on the subbasins (See section 3.2)

d) Sewage - the most readily controllable source of phosphorus - is represented by 120 kg/day in the recreational region all of which is considered as biologically available form.This is about 1/4-th of the total BAP load to the Lake. The proportion of sewage is less important when the total phosphorus form is considered.

e) In respect to the whole Balaton basin the proportion of sewage input to that of other sources mainly of agricultural origin cannot be determined precisely due to the lack of data and to unknown processes (such as the "retention of phosphorus" on the watershed and in the water courses,see later)

Still, it can be estimated that about 50 % of BAP load is of sewage origin (entire catchment), while roughly half of TP is originated from non-point sources of agricultural origin. Thus one can conclude that from the viewpoint of short-term management sewage control while for long-term-management the control of external loads entering the lake via water courses and direct runoff are more effective

2.3. Methods used for the correction of loading figures to account for uncertainties

a) <u>Inflowing streams</u>

The unique set of Zala river data mentioned earlier provided a reliable basis for obtaining more insight into the behaviour of loading rates. This was especially important since the most problematic area of the lake is Keszthely Bay which receives the inputs from the Zala River directly.

On the basis of these data and using the techniques of random sampling, the errors due to infrequent sampling on other streams have been estimated (Somlyódy, 1982) In Table 1. already these corrected loading figures are shown.

One has to note however, that the dynamics of flow in the smaller streams is of far more rapidly fluctuating character than that of Zala River, and the effect of this fluctuation on load conditions is presumable more intense.Thus the 20 % increase of the long-term yearly average load values that was applied to account for unobserved flood events on the basis of Zala data analysis is probably an underestimation of real unobserved effects.

To prove this statement in Figure 3. Zala River annual loads (including all flood events) are compared with the relationships developed for the other stream (virtually not including any flood wave). The good fit between the two independent set of data, indicates that the flood loading rates of smaller streams must be significantly higher than that of the Zala River. Individual flood measurements on pilot areas are also supporting this conclusion. More measurement data on smaller streams are needed to quantify these effects.

b) Sewage load correction

On the basis of relationships developed between flow rate and sewage load and using estimates for per capita sewage output,corrected figures for sewage loading were elaborated. It was shown that sewage data reflects about 60 % of summer peak TP loads. The rest is collected in septic tanks which ultimately causes uncertainties concerning the fate of sewage. This fact is taken into consideration by an increase of load. It was assumed (rather arbitrarily) that 1/4-th of the missing portion, or 15 % of the original summer load estimates belongs to this unidentified category.

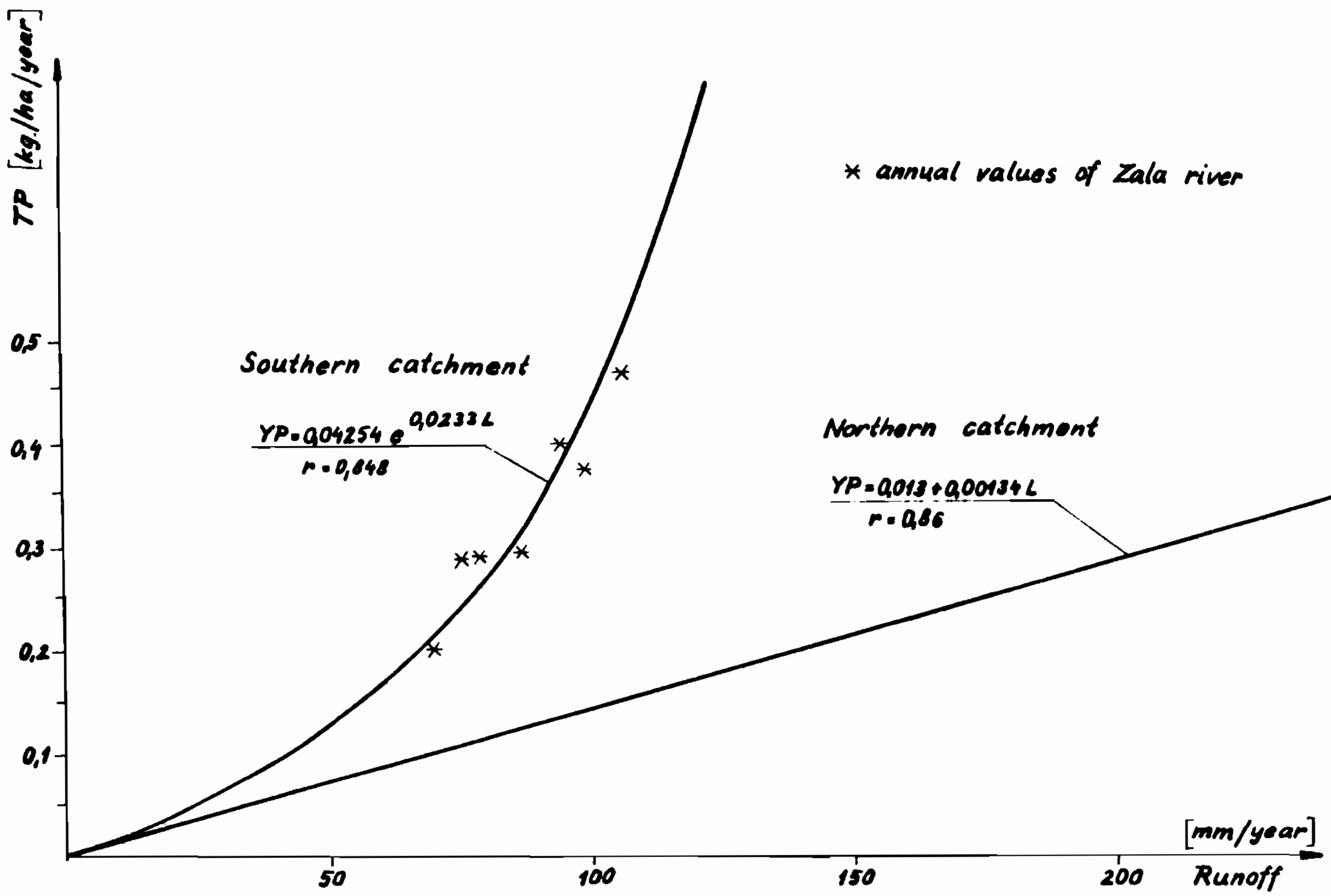


FIGURE 3. TP YIELD vs, RUNOFF RELATIONSHIPS DEVELOPED FOR 20 INFLOWING STREAMS SHOWING ZALA RIVER DATA NOT INCLUDED IN THE REGRESSION

III. NUTRIENT LOAD MODELLING OF LARGE SCALE SYSTEMS

3.1. Introduction

The second objective of this general report is to give a brief account of research achievements that are relevant to the modelling of pollutant load conditions of large water systems, to which latter Lake Balaton case study was an excellent example.

The basic difference of modelling load conditions of large scale systems from that of small "pilot scale" systems is, that for large systems the use of most of the advanced methods and approaches will be out of question due to the lack of data and information in both time and space.

The theoretically acceptable approach would be to route the fate of pollutants through the entire system, as briefly sketched on Fig. 4.

Although such complex computer models are even commercially available in form of program packages; such as HSP, ARM, CREAMS, ACTMO, ASWERS, LANDRUN and others (Anon, 1975, Crawford and Dongian 1974; Free et al. 1975; Novotny et al. 1979) consisting of hydrological, erosion, adsorption, desorption etc. submodels, and are probably applicable for small scale well explored watersheds, their use for larger systems such as Lake Balaton watershed is practically impossible due to the lack of input data and other informations.

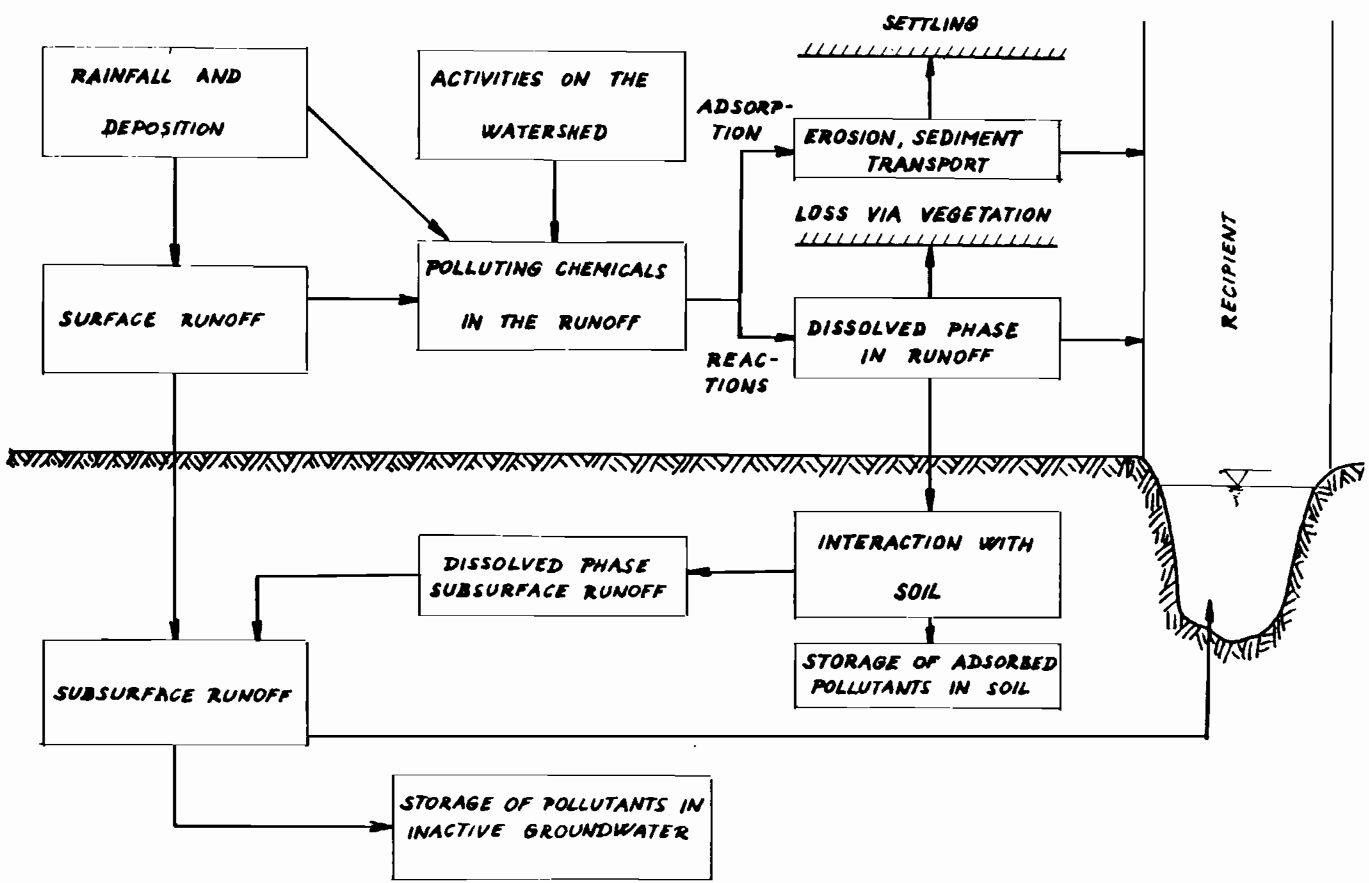


FIGURE 4. MAIN TRANSPORT AND TRANSFORMATION PROCESSES OF POLLUTANTS IN THE LAND RUNOFF

In the frame of the Lake Balaton case study several attempts were made to apply simplified model versions for the estimation of loading rates, with special respect to effects unaccounted for in the available data sets. This attempts are summarized in Table 2.

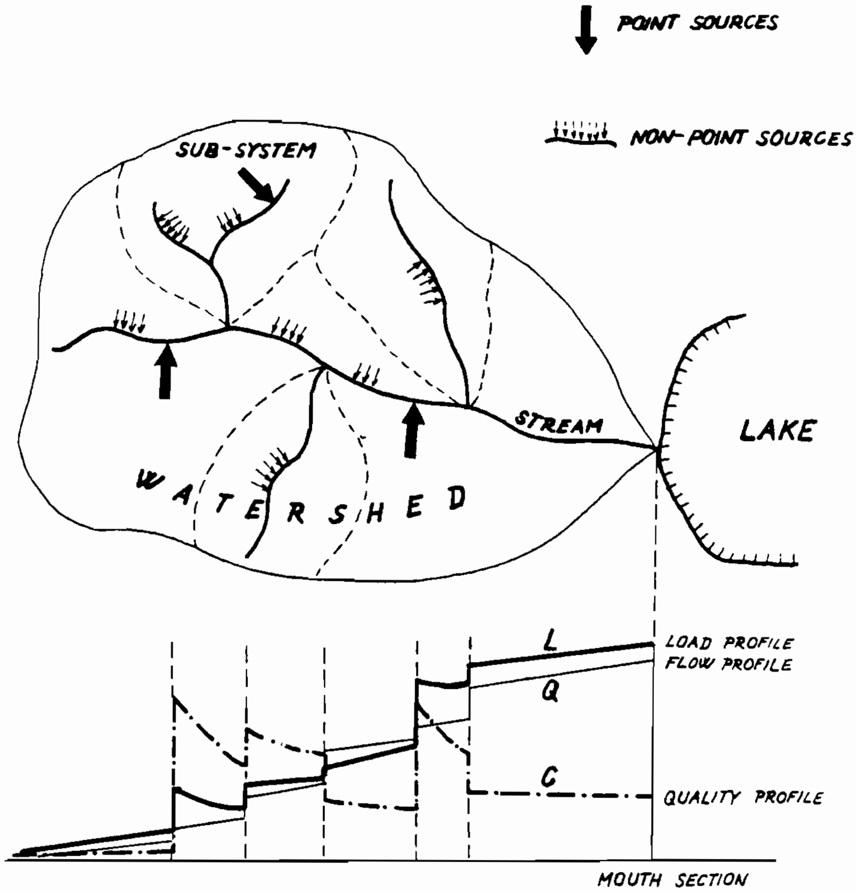
Table 2. seems to confirm our former statement on the applicability of sophisticated models. Although several modelling attempts proved to be useful tools in understanding some of the main governing processes of nutrient load dynamics (Bogárdi and Bolla, 1979; Jolánkai 1982) the application of these models for the total watershed was hindered by lack of data, that could provide for the "missing link" between various submodels.

To elucidate, with some simplification, the essence of this "missing link" let us consider Figure 5. showing a general complex water system with various subbasins and with point and non-point sources of nutrients as well.Even if all inputs-point or non-point - were quantitatively known and/or modelled, the resultant output load at the mouth section of the system would be still unknown. The general framework that would link sub-systems together is missing. This framework could (and should) be provided by a transport and transformation model established for the main recipient stream of the system, that collects transports and transforms all inputs from the watershed.

Table 2

SUMMARY OF PLANT NUTRIENT LOAD DETERMINATION AND MODELLING STUDIES CARRIED OUT IN THE FRAMEWORK OF LAKE BALATON CASE STUDY

References	Brief description of methods used	Study area	Remark
Bogárdi I. and Bolla M. (1980)	Modified Wishmeier-Smith model coupled with SCS runoff model with a stochastic rainfall submodel	Tetves Creek ($70\ km^2$)	Generalization would need much extended data base including sediment phosphorus content and watershed development factors
Joó O. (1979) and Joó and Lotz (1980)	Mass balances with special respect to flood wave vs, base load comparisons; point and non-point source contributions	Zala river ($2622\ km^2$)	Careful and basic examination of the detailed Zala river measurement data
Jolánkai (1976, 1979) Jolánkai and Somlyódi (1982)	Mass balances using all available data sources	Total lake	Analysis of uncertainties was included in the work of Jolánkai and Somlyódy (1982)
Jolánkai (1982a, 1982b)	Unit mass flux response function analogy with Unit Hydrograph , method with the inclusion of preliminary precipitation index	Tetves Creek ($70\ km^2$)	Needs detailed runoff event loading rate data. Generalization is only possible with long records on various characteristic subwatersheds
Beck (1982)	Time series analysis to distinguish phosphorus loadings between point and non-point sources	Zala watershed	Zhe models developed would provide basis for typical TP load series generation
Bolla and Kutas (1982)	Time series and regression analysis used to produce a "mixture of submodels" for Zala river basin nutrient loads	Zala watershed	The model includes also some features of that of Bogárdi-Bolla (1979)



A MODEL THAT TRANSFORMS ALL SUB-SYSTEM INPUTS AND DIRECT INPUTS TO L, C AND Q OUTPUTS AT THE MOUTH SECTION IS NEEDED

FIGURE 5. THE „MISSING LINK" PROBLEM

Although such water quality stream models, considering point and non-point source inputs, are available in theory, their calibration and verification is usually impossible due to the lack of relevant data of water quality profiles along the streams.

In the following three sub-sections first a modelling approach the "Watershed development" approach that bypasses the problem of missing link by directly linking watershed development and natural factors to eutrophication indicators is shown. This method actually bypasses other links too,as it relates the factors affecting nutrient loads directly to eutrophication indicators. Next; a time series analysis on the detailed Zala data obtaining some insight into the general features of the "missing link", will be described in somewhat more details. In the last sub-section longitudinal nutrient profile analysis of Zala River and Tapolca creek will be briefly presented on the basis of field measurements aimed on the determination of natural phosphorus retention along the stream, that is to test a model of the "missing link" type.

3.2 The watershed development approach

The mentioned "bypassing" approach to modeling eutrophication problems in shallow lakes, can be illustrated by approach B on Fig. 6 which directly connects watershed development to the eutrophication of the lakes (Dávid-Telegdi, 1982).

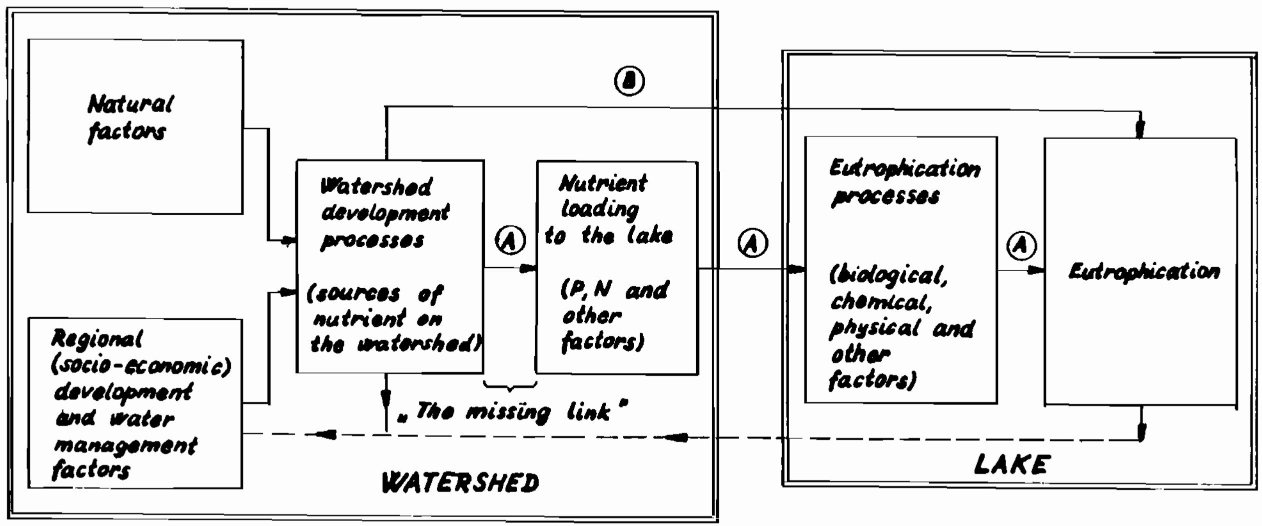


FIGURE 6. APPROACHES (A AND B) TO THE MODELLING OF THE EUTROPHICATION PROBLEM

The formulation of the model rests on the fact that a close connection exists between the eutrophication of a shallow lake and the development of its watershed which generates nutrient loading.

The model is based on the regionalization of the hydrological system of the lake and the connected watershed.In the case of Lake Balaton the model is regionalized into eleven regional units. The lake is considered to consist four water bodies. One of them, the Keszthely Bay is connected to one watershed, while the others are connected to two watershed on each side.

A multicriteria utility function as a watershed development index is used to describe the watershed development in each of the seven watersheds. Based on selected development factors, a system of indicator indices as development criteria was created. It consists 25 indicator indices indicating different elements of regional and water resources development. Among others, the population ratio involved in agriculture and in industry, the visitor loading, the density of point sources, the ratio of arable and forest land use, the fertilizer use, etc. are involved.

The eutrophication is indicated by a moving summer average of chlorophyll-a for each of the four basins of the lake. The eutrophication of a basin depends on the level of watershed development in the contributing watersheds, their position in the hydrological hierarchical system and the eutrophication in the preceding basin. The model is discrete in time on an annual basis which is in harmony with the long--term character of the eutrophication process. The model parameters were elaborated by regression analysis using data for 1975-76.

The watershed development indices are assessed for the seven watersheds of the Balaton basin, for the period 1930-1980.

Although the method cannot improve the load estimate of the lake it has the advantage, namely the usage for analysing the future consequences of control alternatives of the lakes water quality itself. This is illustrated subsequently.

The simulation and forecasting of eutrophication in the lake, are based on three development variations for each watershed for 1981-85, namely a strong, a medium and a zero growth variation. Using different combinations of the specific watershed variations, more than 30 spatial lake protection strategies and their effects on eutrophication were simulated on an annual time scale. Figure 7 illustrates the results of the simulation by a strategy which concentrates the lake protection efforts to the Zala watershed WS_{11} . Therefore in this strategy a zero growth development is considered in the Zala watershed for 1981-85, but in the other six watersheds a strong development variation is considered.

With the help of this simulation the spatial efficiency of the comprehensive control actions in the watershed has

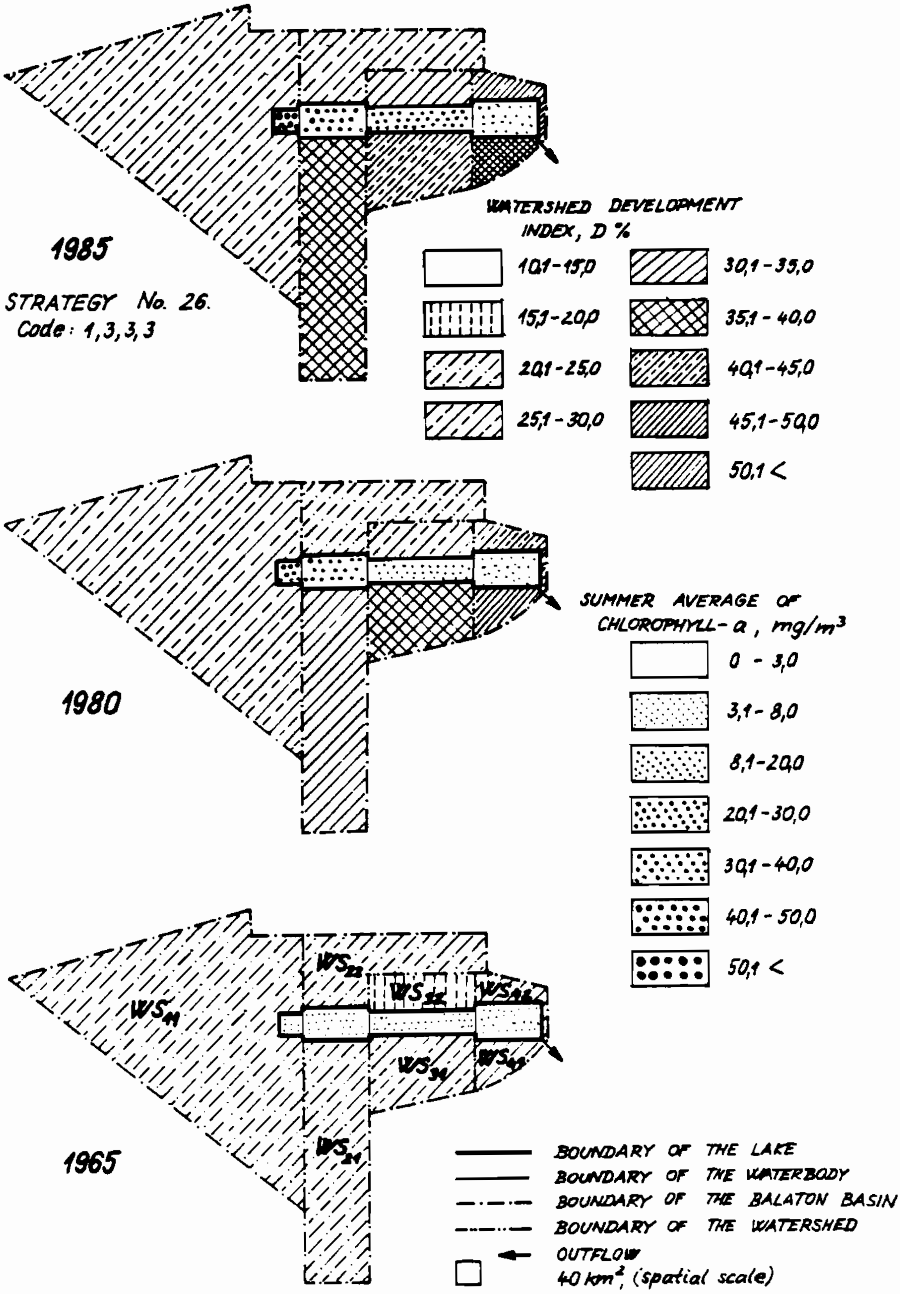


FIGURE 7. THE EUTROPHICATION PROCESS IN THE BALATON BASIN ACCORDING TO STRATEGY NO. 26.

been measured. It indicates that the efficiency of the same volume of protection measures to decrease nutrient loading and eutrophication is 6.2 times greater in the Zala watershed than in the watersheds connected to the Siófok basin; an important conclusion. The corresponding values are 1.7 and 1.1 for the watersheds connected to the Szigliget and the Szemes basin, respectively.

This example clearly shows, that although the approach does not directly solve the problem of missing link because it by-passes this problem, too, it serves as a useful tool for practice. It requires limited amount of data generally available at the beginning stage of such a study and can be used for comparing approximately the effectiveness of various management alternatives.

3.3 Time-series analysis of nutrient loadings in the Zala River

Based on the daily observations Q, SS, TN and TP at two sections at the mouth, Fenékpuszta and 25 km upstream, at Zalaapáti of the River Zala (Joó, 1981) and precipitation data, time-series analysis was performed by using a class of discrete-time multiple input/single output models (Beck 1982). Although the analysis was formulated on a relatively microscopic level of day-to-day changes, the ultimate objectives of the study were relatively macroscopic. For instance, how to distinguish the contribution of point-, and non-point sources in the total load of the lake or how to separate P fractions available and not available for algal growth?

The structure of the model was identified first using the data of 1978 felt to be "typical", while observations of 1975--1977 served for validation. As seen on Fig. 8 (Beck, 1982) the model derived is quite realistic, although peaks are consistently underestimated (also for SS and TN loads); a general feature of such models.

The validation step showed that the model is an acceptable estimator as short-term precipitation events (three or four in a year) are concerned but otherwise the model can be considered rather "invalid" than "valid". For instance, it overestimates the base load of 1976 and 1977, does not work properly in the winter period of 1976-1977 and underestimates the load in July and August, 1975, when a long-lasting flood of slow response took place (the monthly mean TP load was about 30 times larger then the long-term monthly mean). These discrepancies came from the fact, that year 1978 was not really typical; it has the wettest second quarter of the 1975-1978 period and some other events simply did not occur which could have been observed in other years.

From the careful comparison and analysis of observed and simulated results the following conclusions can be drawn:

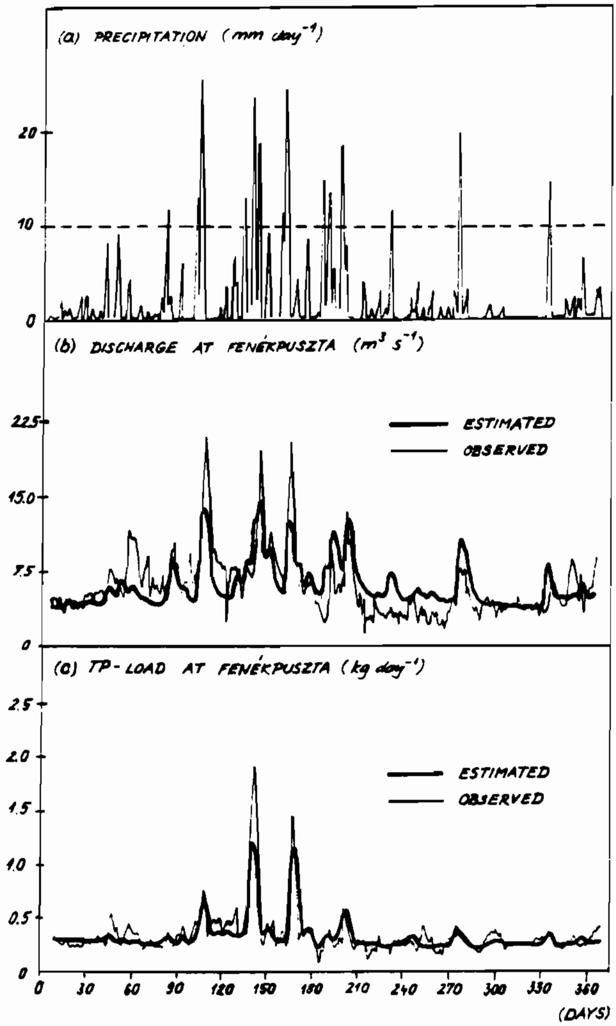


FIGURE 8. OBSERVED PRECIPITATION SEQUENCE FOR 1978 (a); OBSERVED AND ESTIMATED STREAM DISCHARGE (b) AND TP-LOAD AT FENÉKPUSZTA (c)

(i) The yearly average TP base load at the month section ranges between 90 and 140 kg/d. This load is formed by non-particulate P and it is of point source origin, being in good agreement with the estimate of Jolánkai and Somlyódy (1982) (see Section 2).

(ii) The TP load arising from particulate material are strongly dependent upon precipitation. The long-term average of this P load is about 40-60 kg/d, a relatively low value. This is mainly due to in-stream processes. Namely, the analysis performed suggests that below a threshold discharge value of 10-12 m^3/s particulate matter is deposited downstream Zalaapáti. The comparison of records for Zalaapáti and Fenékpuszta shows that spatial changes in the particulate P load are determined by deposition and scouring but not by soil erosion from the watershed. Erosion and transport of eroded soil to the river is important upstream Zalaapáti.

(iii) The precipitation induced non-particulate P load (probably of agricultural origin) has a surprisingly high value, about 30 kg/d in an average. TP originating from the watershed between Zalaapáti and Fenékpuszta are predominantly of agricultural origin and composed largely of non-particulate fractions.

(iv) It is evident from the analysis that we have a lack of knowledge as seasonal changes (evapotranspiration, snowmelt, flooding and slow drainage of marshland located at the mouth of the river, variability of treatment plant performance, in--river processes etc.) are considered and thus it is very difficult to derive a load estimate for subsequent years different in character.

The conclusions drawn contribute to the resolution of the missing-link problem on one hand, and indicates future research needs, on the other hand.

3.4 Phosphorus profile studies of Zala River and Tapolca creek

As mentioned earlier in this section the calibration of stream models that could give account of the longitudinal variation of phosphorus loading rates while considering both point and non-point source inputs would be of outstanding importance, in that such models could provide a general tool for linking the outputs of sub-basin models together thus serving for the elimination of the "missing link" problem.

In the framework of this research program among others two field measurement series were carried out:

a) Detailed field measurements upstream and downstream of the Tapolca sewage treatment plant's effluent outfall, on the Tapolca creek, (Fig. 2.) aimed on the calibration of the stream model, on the estimation of retention rates, and finally on the assessment of the effects of an assumed P removal at this treatment plant.

b) Longitudinal profile study of the Zala River (Fig. 2) between Zalaegerszeg and the mouth section, with frequent measurements of four stream sections and the main sewage discharge (Zalaegerszeg STP) and sampling and discharge measurements on all of the 32 inflowing tributaries.

Several model forms have been examined for suitability to describe measured nutrient load profiles along the stream. The two main requirements for the model were as follows:

a) It should be able to consider point and non-point source inputs as well, and

b) It should be able to account for the natural phosphorus retention (settling, uptake by plants, and other losses) in the stream bed.

Finally a steady-state model assuming a first order loss term was selected.

In words this model states, that the change of mass flux (load) within an incremental length of the stream is equal to the non-point source input minus the retention of the constituent in the stream bed. Concerning this latter term it is assumed that retention is directly proportional to the concentration and to the cross-section area with a proportionality factor K.

Some of the results of mass balance calculations and model fitting experiments are summarized in Table 3.

Table 3.

Stream / Constituent	Specific loss calculated from mass balance in percentage of the total input mass flux %/km	Retention coefficient K as calibrated with the model day^{-1}
Zala / Total-P	0,72-1,89	0,29-1,1
Tapolca / PO_4-P	1,49	0,88

The results of the two streams were similar and allowed some generalizable conclusions as far as further extensions and practical use of such models are concerned.

The model can be well fitted to the observed load profile data and in the case of Tapolca creek studies there were some evidences indicating the simulation capability of the model.

The natural retention rates of phosphorus are within a seemingly well definiable range.

Both, in the Zala River and Tapolca creek case it was apparent, that natural retention effects in reducing loading rates are much smaller than the removal that could be achieved by removing phosphorus at the treatment plants.

Although these preliminary conclusions should be justified on the basis of more extensive studies, the results are meaningful and promising, and suggest the general applicability of this modelling approach.

Models of this type will be most likely able to provide the "link" between various submodels developed for the various sub-watersheds of the system.

IV. SUMMARY

The external nutrient loads to Lake Balaton have been determined making use of available monitoring data and relevant other informations. Due to relatively infrequent observations of both surface and sewage discharges, long-term average loading figures have been established (1975-1981), based on raw data and then corrected to account for uncertainties resulting from hydrological variations and infrequent sampling.

Nutrient load modelling attempts have been reviewed together with those of the relevant literature of this field. It has been concluded that up to now no modelling approach can claim the general applicability for the total Balaton watershed (or similar watersheds) due to the fact that suggested model structures will not meet the data availability of this system.

In order to achieve a more generalizable description of nutrient load conditions of the entire lake-watershed system three approaches have been discussed.

The final conclusions may be formulated as follows:

a) Acceptable nutrient input loading figures were provided for Lake Balaton and its basins enabling also some insight into the uncertainties associated

b) These figures can now be the basis of
   - Lake ecological model investigations and
   - Water quality management modelling studies

c) These results provide also starting point for further research, namely
   - More work should be focussed on the identification of non-point sources and on factors affecting the fate of pollutants originating from these sources (i.e. storm runoff events)
   - More observations on point sources are required
   - Stream water quality models (load profile studies) to link sub-system inputs together and transform them to lake inputs, are needed.

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SEDIMENTS AND SEDIMENT–WATER INTERACTION

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Introduction

This contribution will deal with the following subjects:

1. P-accumulation and concentration in sediments.
2. Chemical speciation of P and transformations in sediments.
3. Release mechanisms, differentiating between particulate and dissolved P.
4. Internal loading; its magnitude and its role in the nutrient cycle.
5. Effects of management measures upon sediment behaviour.

After a short general discussion of each of these themes some relevant information pertaining to Lake Balaton will be reviewed. It should be kept in mind when considering the role of sediments in the eutrophication process, that two levels of resolution can be discerned:

- a coarse picture, based on lake wide average concentrations in a well-mixed top layer of sediment and temporal variations covering years or decades with the objective to understand and to model long term trends and the effectiveness of management measures;

- a detailed picture, taking into account horizontal and vertical gradients in sediment characteristics, daily or seasonal variations in concentration profiles, redox conditions and fluxes; with the main objective to understand and/or to model the cycling of nutrients in relation to the algal growth. This paper falls into the first category.

Accumulation and Concentration

The accumulation is the quantity or percentage of a substance - in this case phosphorus - retained in a lake. The retention of phosphorus in freshwater lakes is very often more than 50 % of the incoming load; lake Balaton has a very high retention coefficient of approximately 95 %, corresponding to a net accumulation rate of about 0.7 g P/m^2, year. This is a lake wide, annual average; in Keszthely Bay the accumulation rate is higher. A temporal variation is related to rainfall, the run-off and concomitant variation in suspended solids in the tributaries. Also the field data suffer from uncertainty.

According to Vollenweiders (1979) criteria and OECD (1982) studies a loading of 0.7 g P/m^2, year at a hydraulic residence time of about 2 years results in average chlorophyl concentrations of about 20 mg/m^3 (90 % conficence limits 8-70 mg/m^3). For Keszthely Bay the corresponding figures are roughly 3.5 g P/m^2, year, τ_w = 1 year and chlorophyl = 60 mg/m^3. Predicted average in-lake phosphate concentrations on the basis of these data are about 100 mg/m^3 (lake wide) and 250 mg/m^3 (Keszthely), actual figures are roughly 50 % of these values,

but the spatial variation in the loading is reflected very pronounced in the longitudinal variation in chlorophyl and phosphate concentration. This will also affect the sediments, but here the high buffering capacity of the bottom material for phosphate is responsible for a very slow response. The build up of a phosphate concentration in sediments can be envisaged to proceed as outlined in figure 1.
A net quantity of phosphate accumulates annually in the sediments ($S = 0.7$ g/m², y) together with other material which settles and causes an annual deposition ΔL which is generally in the order of a few mm per year. This material is mixed through the top layer of the sediments by repeated resuspension and sedimentation but particularly by the activity of the bottom fauna. The mixing depth (and intensity) may vary considerably, but an order of magnitude is about 10 cm. In the case of phosphate decay will be absent and a mass balance over the mixed layer results upon integration in:

$$C = \left\{ C_o - \frac{S}{\Delta L} \right\} \exp - \frac{\Delta L}{L} \cdot t + \frac{S}{\Delta L} \qquad (1)$$

in which

C = phosphate concentration in the sediment. (g/m³)

C_o = phosphate concentration at time zero.

The equilibrium concentration is $S/\Delta L$, the rate at which a new equilibrium will be attained is controlled by $\Delta L/L$. As this value is rather small, the time constant of this sediment dilution rate is in the order of decades. This means that the onset of high loading rates in the sixties has not yet resulted in equilibrium concentrations in the sediment. The reverse is that a reduction in present day loading will also cause a slow exponential decrease in the sediment concentration

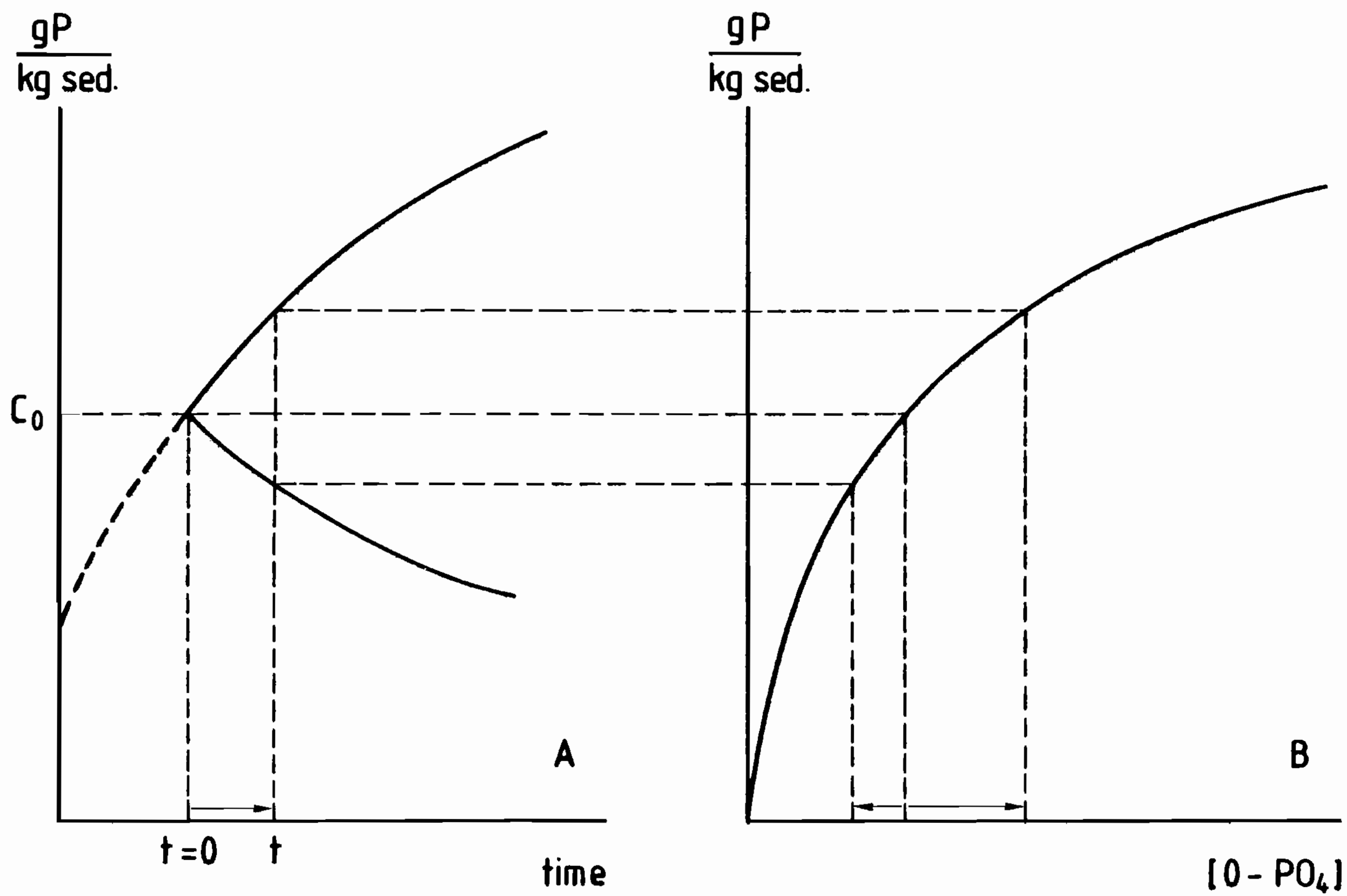

Figure 1 Effect of step change in external phosphorus loading on sediment concentration in mixed layer (A) and on pore water equilibrium concentration of o-phosphate (B).

(see figure 1). However, because this decrease is non-linear and there is also a non linear relation between the sediment-P concentration and the equilibrium concentration in the interstitial water in contact with the sediment, a decrease in loading can be magnified considerably in the concomitant diffusive flux out of the sediment into the overlying water. (figure 1).

This general picture of accumulation and concentration is complicated by the inhomogeneous lake bottom, the spatial variation of the accumulation rate due to internal transport of sediment caused by erosion and deposition, the spatial variation in sediment mixing depth and mixing intensity due to the selection of favourable environments by the bottom fauna etc. Essentially the required information for the description and prediction of sediment composition on this basis is presently not available for lake Balaton. However, other approaches allow some quantitative estimates to be made.

For Keszthely Bay the annual P-input was estimated at about 100-150 tons, whereas the river Zala carries about 15 à 20.10^3 tons of suspended solids into the Bay. Assuming that for the greater part both P and suspended solids accumulate, this would result in new sediment containing about 7 mg P/g sediment. Because the sediments in the Bay contain ± 65 % $CaCO_3$ and these carbonates are formed within the Bay by the photosynthetic activity, the concentration would be reduced to about one third of this value, or about 2.3 mg P/g sediment. However, the present content of $CaCO_3$ in the sediment is probably lower than in the accumulating material due to the enhanced eutrophication in recent years which stimulates the formation of $CaCO_3$. When the sedimenting material contains 80 % of calciumcarbonate, the phosphate content will be 1.4 mg/g sediment.

The actual $CaCO_3$ content should be estimated from sediment trap data. The present concentration of phophorus in the sediments in Keszthely Bay is 0.7 mg P/g sediment. This v lue suggest that the sediment is not yet in equilibrium with the sedimenting material but that a gradual enrichment is taking place. The approach presented here is complicated by the fact that part of the accumulating carbonates are dissolved in the sediments by carbon dioxide produced by mineralisation.
Lake wide roughly 420 tons of P accumulate and a mass balance on calcium indicates a retention of about 34.10^3 tons of this element. This would result in a P:$CaCO_3$ ratio of 1:200 or 5 mg P/g sediment. Taking into account the percentage of $CaCO_3$ in the sediments (± 50 %) and the settling material (probably higher) and also the contribution of Mg to the carbonate precipitation, the concentration in the sedimenting material will be in the order of 2-3 mg P/g sediment, but certainly higher than the actual concentration in the deposits (0.4-0.7 mg P/g sediment). Again the conclusion is that a gradual enrichment of the sediments with P is in progress.

Chemical speciation; Transformations

The forms of phosphorus present in the sedimenting material, the water and pore water and in the sediments itself are of a wide diversity. Certain compounds are formed within the lake (autochthonous), other forms are derived mainly form eroding material in the watershed (allochthonous). Table 1 summarizes the main forms present in the particulate sedimenting material.

Allochthonous		Autochthonous	
- Absorbed on clays	*	- Absorbed on $CaCO_3$	*
- Associated with (hydr)oxides of Fe, Mn, Al ...	*	- Ca-phosphates	
- Salts of Ca, Mg, ...		- Associated with oxides	
- Organic debris	*	- Detritus, dead algae	*
- Minerals			

Table 1. Chemical forms of phosphate in settling material in lakes. * indicates species identified in the case of lake Balaton or almost certainly present.

Both the lake water and the pore water contain ortho-phosphate as well as organic phosphates. In the sediments transformations occur in which the dissolved ortho-phosphate plays an important role; see table 2.

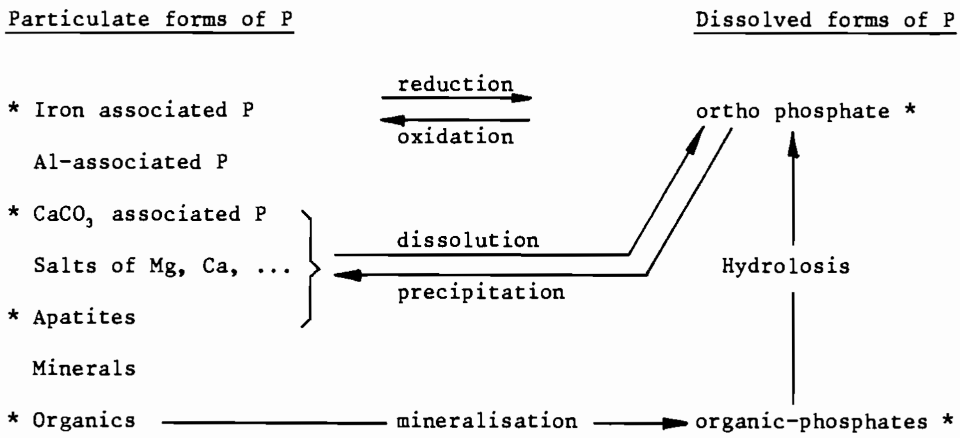

Table 2. Main chemical forms of phosphorus in sediments and major pathways of P-cycling. * = forms indentified or almost certainly present in lake Balaton sediments.

The tables 1 and 2 indicate that the phoshpate chemistry and cycling is rather complex. Because the species present are often non-homogeneous and amorphous or of poorly defined nature, prediction of the solubility and equilibria are generally difficult. The transformations indicated in table 2 depend on local environmental conditions such as redox potential and pH. Due to spatial and seasonal variations related to sediment composition, temperature, oxygen supply etc., the concentration of dissolved ortho phosphate in the pore water may vary strongly. This is important for the release of P and the overall cycling of the element.

Release mechanisms

It is useful to discern between the transport of particulate-P and dissolved P from the sediments into overlying water.

Particulate phosphorous

Resuspension of sediment particles followed by desorption of the associated phosphates can contribute to the lake water phosphate pool. The magnitude of this flux will depend on the rate of resuspension and on the adsorption/desorption characteristics of the particles in relation to the environmental conditions of pH, redox potential and actual phosphate concentration. The rate of resuspension is controlled by the properties of the sediment (particle size distribution, water content, cohesiveness, specific weight, bottom roughness) and the shear stresses near the bottom. The properties of the sediment can change appreciably by the activities of macro-invertebrates feeding on the sediments. One of their effects can be the deposition of fine material on top of the sediment after ingestion and digestion of selected food from deeper strata. The shear stresses can be induced by flow or by waves (figure 2), hence in lakes they are predominantly a function of the wind field. Waves cause a depth dependent horizontal velocity, the maximum value thereof can be shown to be:

$$U_{D,max} = \frac{\pi H}{T.\sinh(2\pi D/L_D)} \tag{2}$$

in which H = wave height (m)

T = wave period (s)

L_D= wave length for shallow lakes (m)

For H, T and L_D emperical relations have been assessed which take into account the fetch.

Equation (2) assumes steady state conditions. Lam and Jaquet (1976) postulated a resuspension flux proportional to the difference between $U_{D,max}$ and some critical minimal velocity. As an alternative the production of potential energy associated with the resuspension flux can be set equal to the dissipation of wave energy (neglecting dissipation into heat) which results in a proportionality of the rate of resuspension with $U_{D,max}$ 1.875. Calculation of the vertical flow field caused by wind results in bottom frictions (Banks, 1975) which, when substituted in an energy dissipation equation, also results in some relation between the rate of resuspension and wind velocity. The actual concentration in the overlying water is the result of both resuspension and sedimentation. The latter is generally supposed to be proportional to the concentration of suspended material. The proportionality constant will be approximately equal to the settling velocity for larger particles, but for small

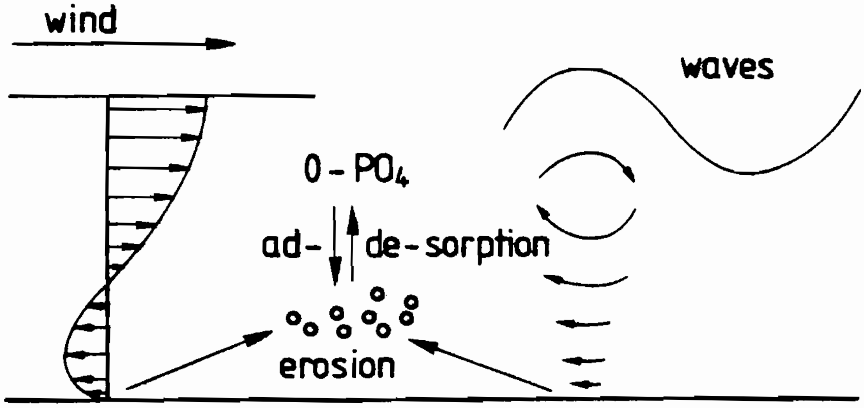

Figure 2 Mechanisms of wind-induced erosion.

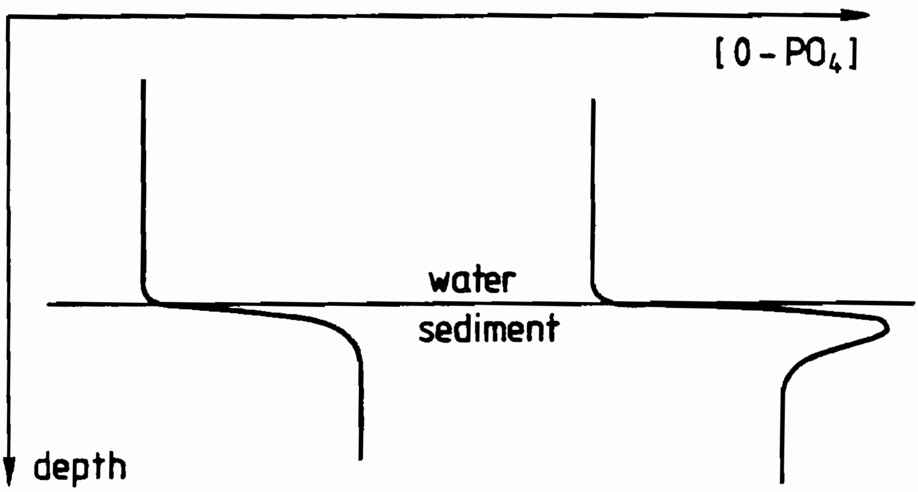

Figure 3 Concentration profiles of dissolved o-phosphate near the water-sediment interface.

particles diffusion in the boundary layer is predominant (Lick, 1982).

The theoretical models described briefly in the foregoing contain several unknown coefficients, apart from the uncertainties in the assumptions used. Hence an approach must be chosen for the assessment of these parameters. In practice two methods are used. The first is an analysis of time series of suspended solids concentration as a function of wind velocity and application of a parameter estimation technique which seeks to minimize the difference between model predictions and field data. This metholology has been applied in the case of lake Balaton and will be referred to in the following section. A second approach is to measure the actual relation between shear stress and rate of resuspension for the sediments under study and calculate the bottom stress from a hydrodynamic model. This method has been applied by Fukuda and Lick (1980) and Sheng and Lick (1979).

Dissolved phosphates

The release of dissolved phosphorous compounds, mainly orthophosphate, is generally controlled by diffusion in the boundary layer of the sediment. Sometimes advection caused by groundwater flow affects the transport; nearly always interactions between the solute and the solid phase have an important influence upon the concentration profile in the pore water near the watersediment interface. The differential equations describing the phosphate concentration is

$$\frac{\delta c}{\delta t} = - u \frac{\delta c}{\delta z} + D_{eff} \frac{\delta^2 c}{\delta z^2} \pm R(z) \qquad (3)$$

in which

c = phosphate concentration (mg/m^3)

u = advective velocity (m/s)

D_{eff} = effective diffusivity (m^2/s)

R = rate of production or removal of phosphate ($mg/m^3.s$)

z = depth coordinate (m)

t = time (s)

The solution of this equation, either analytical or numerical, is hampered mainly by the complex chemistry which is covered by the term R. It includes the rate of phosphate production by mineralisation of organic matter and the desorption and adsorption processes on clays, oxides and carbonates which are dependent on pH and redox conditions. During certain periods of the year the input of detritus into the sediments in productive lakes can lead to an accumulation of this material in the top layer. This leads to a higher local production rate of phosphate by the fast mineralisation of this fresh material. The resulting concentration profile may take the form shown in the right hand side of figure 3. Analysis of such a situation on the basis of a depth averaged pore water concentration can lead to a serious underestimation of the diffusive flux into the overlying water. The activity of the bottom fauna will counteract a stratification in sediments.

In most sediments the annual variation in redox conditions, caused by increased oxygen consumption as a result of enhanced productivity and high temperatures during the summer, affect strongly the phosphate release. This is due to the reduction of iron-hydroxo-phosphates and the subsequent dissolution of phosphates (Lijklema, 1980).

Generally the prediction of diffusive fluxes cannot yet be based on a deterministic model, but relies on field an laboratory measurements under varying environmental conditions.

Magnitude of internal loading in lake Balaton; comparison with other fluxes

In this section an estimate of fluxes in the phosphate cycle in lake Balaton will be presented.

Resuspension and desorption

Somlyódy (1980) has applied Kalman filtering techniques for the assessment of the parameters in a simple hydrodynamical model for resuspension. The result is a prediction of the rate of resuspension:

$$\phi_{resusp.} = 0.034\ W \tag{4}$$

in which

W is the wind velocity (m/s)

ϕ is the flux in $kg/m^2.day$.

Equation (4) does not take into account spatial variations in W, in water depth or sediment characteristics. Nevertheless a reasonable fit with the field data is obtained. The desorption of phosphate from resuspended sediment has been studied by Gelencsér et al (1982).

The rate of equilibration has not yet been the object of a detailed study, but preliminary results indicate at least fairly rapid equilibration (within 30 minutes). At the prevailing lake water concentrations (1-4 μg PO_4-P/l) and suspended solids concentration of 50-1000 mg/l the desorption was generally between 5 and 10 μg P per g sediment. Combined with equation (4) this results in a flux of 0.8-1.7 mg P/m²,day at an average wind velocity of 5 m/s.

Diffusion

No measurements are available of the diffusive flux out of the sediments. Also accurate data on pore water concentrations and particularly of concentration profiles in the sediments are not available. Hence only an order of magnitude can be predicted. Based on a simple Fick's law approach:

$$\phi = D_{eff.} \frac{\Delta c}{\Delta z} \qquad (5)$$

with $D_{eff.} \simeq 10^{-9}$ m²/s

$c \simeq$ 50-250 μg/l

$z \simeq$ 2 mm - 2 cm

the diffusive flux will be somewhere in the range of 0.2-10 mg P/m²,day. In eutrophic lakes release rates of 50 mg P/m²,day are not an exception; hence the higher figure in the range is probably rather realistic for lake Balaton, but field data are needed urgently.

Primary Production; Mineralisation

Productivities measured in lake Balaton vary between low values of 0.05 g C/m^2,day to several grams per m^2 and per day. Average summer values are in the order of 1 g C/m^2,day with a tendency to higher values in Keszthely Bay. Assuming a C:P ratio of 50:1 in algae, this means a flux of phosphate of about 20 mg P/m^2,day from the dissolved o-phosphate pool into the biomass.

There are indications that a large fraction of the assimilated carbon is mineralised in the water; this is mainly gathered from bacterial counts. This means that the phosphate flux from biomass to the o-phosphate pool is also in the order of 20 mg P/m^2,day during the summer, but on the average certainly smaller than the assimilation rate because there is a net accumulation of organic matter in the sediments and a part of the mineralisation occurs there. Also assimilation and mineralisation will be out of phase with the mineralisation showing a certain delay.

Settling, lime precipitation

The total rate of settling of phosphorous compounds is unknown, but certainly about 50 % of the external loading (ca 2 mg P/m^2,day) will settle because it is in an insoluble form. The flux of phosphate associated with biomass and detritus however is unknown; the magnitude may be about 30 % of the quantity assimilated.

An important potential mechanism for the removal of phosphate from the water body is co-precipitation with lime. Lake Balaton is a hard water lake and as a rule the water is saturated with $Ca(Mg)CO_3$. Hence each mole of CO_2 consumed by algae can result in one mole of $CaCO_3$ formed:

$$2\ HCO_3^- + Ca^{2+} \rightarrow CaCO_3\downarrow + CO_2 \qquad (6)$$

At a daily production of 1 C/m² and assuming 70 % mineralisation in the water column, this would result in the formation of 2.500 mg $CaCO_3$ per m²,day. Freshly precipitating $CaCO_3$ can absorb phosphates very effectively; a P:$CaCO_3$ of 1:100 at high o-phosphate concentrations or 1:1000 at low o-phosphate concentrations is a typical range. This corresponds to a flux of ± 2.5 mg P/m²,day, because normally the phosphate concentration is low or very low. Again, this is a very rough estimate, which should be substantiated by field data.

Putting these fluxes together the picture of figure 4 is obtained. Although the numbers in this flow sheet are not at all accurate and vary in time and space, certain conclusions are obvious. The main conclusion for this contribution is that internal loading contributes considerably to the productivity of the lake and is probably more important for the actual productivity than the external loading. Yet the management should focus on the external loading and therefore the influence of the external loading on the internal loading will be discussed in the following section. It is worthwile to note here that the fairly constant o-phosphate concentration, both within 24 hours and a year, can be explained by compensation mechanisms: low o-phosphate concentrations will tend to stimulate desorption and diffusion, whereas high phosphate concentrations will stimulate uptake by algae and co-precipitation with lime.

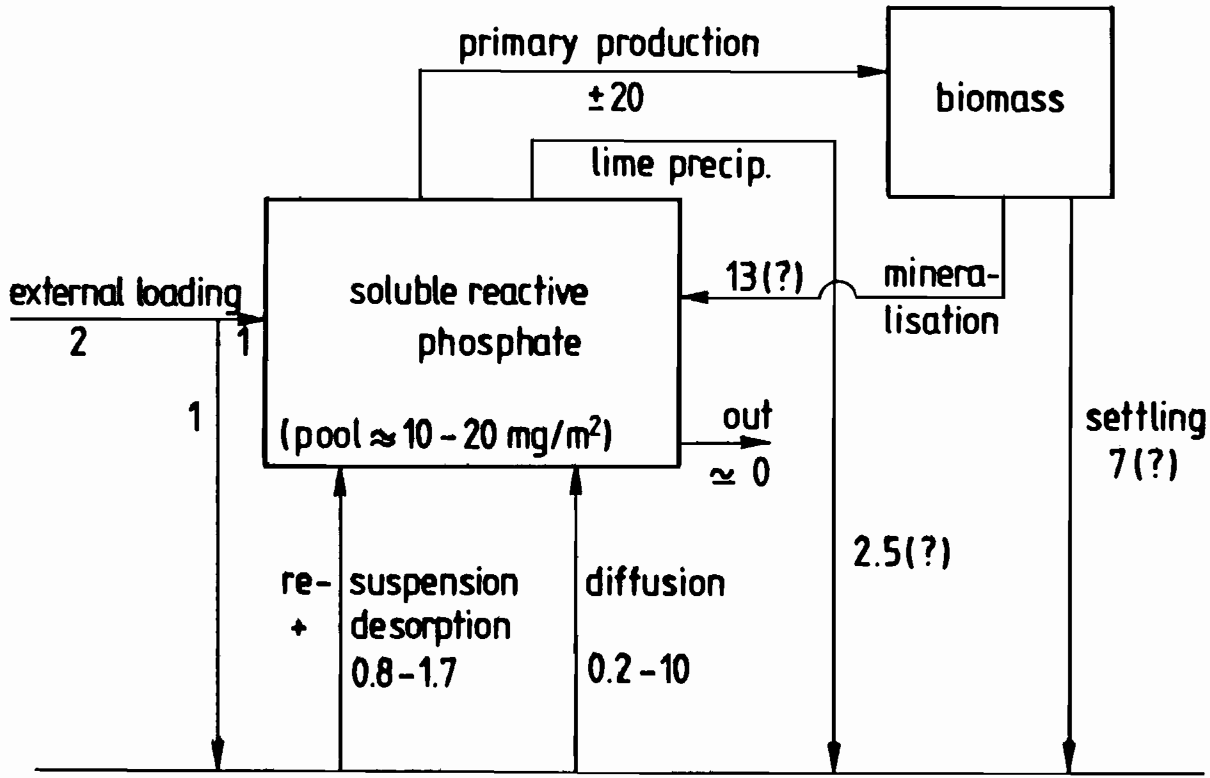

Figure 4 Orders of magnitude of phosphate fluxes in lake Balaton.

Units: mg P/m², day.

This is an important argument to study productivity and eutrophication from the viewpoint of fluxes and rates rather than focussing on concentrations, which are the net result of balancing processes.

Management and its effects upon sediments

The main instrument to control the productivity of lake Balaton is reduction of the nutrient input; probably phosphorus is the most promising element to tackle. Disregarding the transformations in the water body (figure 4), the net result is that the greater part of the 2 mg P/m^2, day reaching the lake, accumulates in the sediments. The soluble and readily available fraction thereof is the most harmful, because it can continue to cycle through the system until it is lost through the outflow or insolubilised and buried in the sediments. Phosphates associated with suspended solids and settling into the sediments can also become available by desorption and diffusion or resuspension and desorption, but these processes have at least one barrier because transport is required. This transport, especially the diffusive transport, is controlled by concentration gradients. Returning to figure 1, it will be clear that a reduction of the phosphate loading of the sediment will change the adsorption-desorption equilibria. At low phosphate concentrations mainly the high energy adsorption sites in the sediment will be occupied and the equilibrium concentration in the liquid is low. The combined effect of the exponential response of the sediment concentration upon a reduced input and the non-linear adsorption isotherm cause a much faster reaction of the equilibrium phosphate concentration in the pore water than corresponds with the slow sediment dilution rate. This in turn affects the internal loading and within a few years the trend in the eutrophication can be reversed and manifestitself in a decreasing productivity. A further coupling which may accelerate this restoration is the reduced loading of the sediments with organic matter. This will reduce the extent and time during which anaerobic conditions can prevail at or near the sediment-water interface. It is well known that such conditions stimulate the release of phosphate from the sediments. The whole argument is that the historical process of eutrophication can be reversed by returning to the original conditions with respect to phosphate inputs. In summary:

- The effect of a reduction in external loading will be, depending on the magnitude of this reduction:

    a. Slackening of the rate of accumulation in the sediments or a reversal of the trend.

    b. Stabilisation or reduction of the internal loading.

    c. Stabilisation or reduction of primary productivity.

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BIOCHEMICAL PROCESSES IN LAKE BALATON

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The relative importance of metabolic activities in the littoral, pelagic and benthic zones differ from lake to lake. In shallow waters, like Lake Balaton the benthic zone is of higher importance, than in the deep ones, as the surface to volume ratio is inversely related to water depth. The littoral zone is also typically larger in shallow waters, but Lake Balaton is an exception in this respect. Here the distribution of the macrophytes is restricted to a narrow belt by the very strong wave action, and more than 95 per cent of the surface remains open water. Accordingly the dynamics of the water is important not only in the redistribution of the nutrients between the basins and between the water and the sediment, but actually it is due to these waves, that despite of a mean depth of 3.2 m, Lake Balaton has the appearance and many characteristics of a real lake. On the other hand, the waves considerably increase the importance of the sediment in the metabolism and eutrophication of the lake. Due to this significant role the sediment/water interaction is the subject of a separate general report, this paper therefore will center on the main processes of eutrophication in the open water.

The composition, quantity and photosynthetic activity of the phytoplankton

Quantitative phytoplankton studies have been initiated by Olga Sebestyén already in the thirties. Between 1936 and 1951 she counted the cell numbers and by the cell volume method determined the biomass of the Dinoflagellatae. A definite increase in the amount of Ceratium hirundinella, the dominant algal species of this lake was demonstrated /Sebestyén 1953, 1954/. In 1945, 1947, 1949 and 1951 the cell number and the biomass of the other algal groups were also determined from samples collected at monthly intervals in Basin IV in front of the Limnological Institute /Tamás 1955/. From the data of the two authors the biomass of the total phytoplankton can be reconstructed. Its annual mean was about 0.3 mg fresh weight/litre in the forties and 1.2 mg fresh weight/litre in 1951, the latter being an exceptionally high value. In these early studies the Kolkwitz chamber was used, therefore the smaller algae were not detected. The cell number of the plankton is therefore strongly underestimated, but concerning the biomass the error is probably less than 20 per cent.

In 1965, 1966, 1967, 1974 and 1976 samples were collected monthly or fortnightly from all the four basins of the lake, and they were already counted under the Utermöhl microscope /Tamás 1974, 1975, Vörös 1980/. Similar cell number and biomass studies paralleled the primary production measurements in the four basins in the seventies /Herodek and Tamás 1976, 1978, Herodek et al. 1982/. In 1977 weekly in 1978 bi-weekly samples were collected for such algological studies from the Basin I and Basin IV /Vörös 1982/. The altogether more than 20 000 data were analysed by computer.

The winter, early spring, late spring, summer and autumn phytoplankton showed well separated clusters /Vörös and Kiss 1983/. Under the ice there are small, slowly sedimenting and some actively moving species. The strong outburst of diatoms starts usually a few weeks after ice break. For May the nutrients are depleted from the water, the biomass of the phytoplankton diminishes and transient communities with rapidly changing structures can be observed. The development of the summer phytoplankton starts in the middle of June, and attains its maximum in July or August. The autumn community appears around mid--September after the fall of the temperature, and its biomass remains low.

During the eutrophication of the lake the structure of the phytoplankton showed definite changes. In the spring community earlier the disc shaped species /Centrales/, mainly Cyclotella bodanica and Cyclotella ocellata prevailed, but from the middle of the seventies the thin, needle like /Pennales/ Synedra acus and Nitzschia acicularis are most abundant. The changes in the summer community are even more expressed. Originally the whole lake was dominated by Ceratium hirundinella. In the most polluted parts of the lake the blue-greens appeared from time to time in larger amounts already since 1965, and from 1973 the water blooms of the filamentous, heterocystic blue-green species /Aphanizomenon flos-aquae, Anabaena spiroides/ became a regular phenomenon. In the larger part of the lake /Basin III and IV/ the amount of the blue-greens remained low until recently, but in the summer of 1982 already the whole lake was invaded by a filamentous blue-green species, Anabaenopsis raciborskii.

The biomass of the phytoplankton in Basin I /Fig. 1/ was always lower than 10 g/m^3 in the sixties. In 1973, when the very high primary production was found /see later/, the

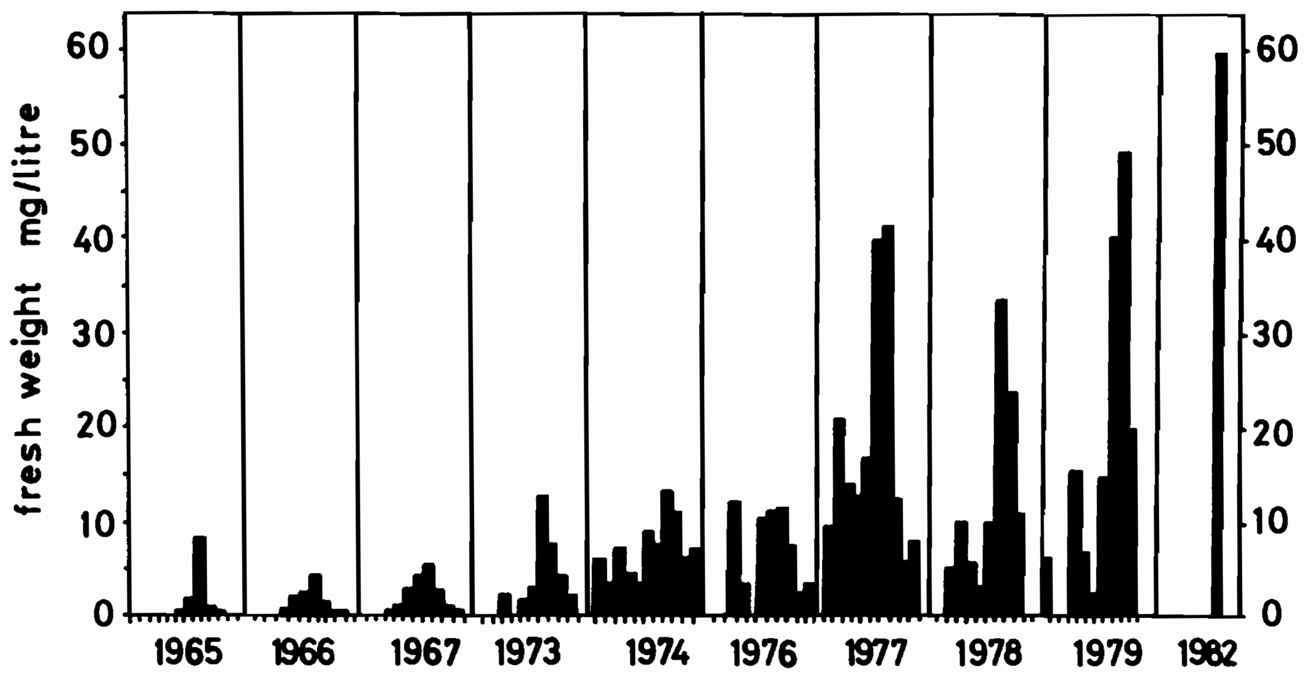


Figure 1. The phytoplankton biomass in Basin I /Vörös 1982/

biomass attained 13 mg/litre. Similar values were measured in 1974 and 1976 too. In 1977 a second big leap took place with a phytoplankton peak higher than 40 mg/litre. In 1978 the summer maximum was somewhat lower, due to the cold weather in this year. In 1979 50 mg/litre, in 1982 60 mg/litre were the biomass maxima.

The increase of the phytoplankton biomass is demonstrable in all sampling stations /Fig. 2/. At Tihany /Basin IV/ the highest value was only 1 mg/litre in 1965, the summer peak of 1978 was already 8 mg/litre, while during the Anabaenopsis raciborskii bloom of 1982 14 mg/litre was the concentration of the algae in the water.

Because species with smaller cell volume became dominant, the number of individuals increased much more than the biomass. At Tihany the number of algae was only a few hundred individuals per ml in the sixties, more thousand in the seventies and more than ten thousands in 1982 /Fig. 3/ /Vörös 1982/.

Cell counting and the determination of the biomass by the cell volume method are rather time consuming. The a-chlorophyll measurement is much easier, and therefore it became in general use as an index of the amount of algae. This way however the information contained by the composition of the phytoplankton is lost, and it must be noted, that rather different chlorophyll/biomass ratios are published. In Lake

Figure 2. The maxima of phytoplankton biomass in August-September 1965, 1978 and 1982 /Vörös, unpubl./

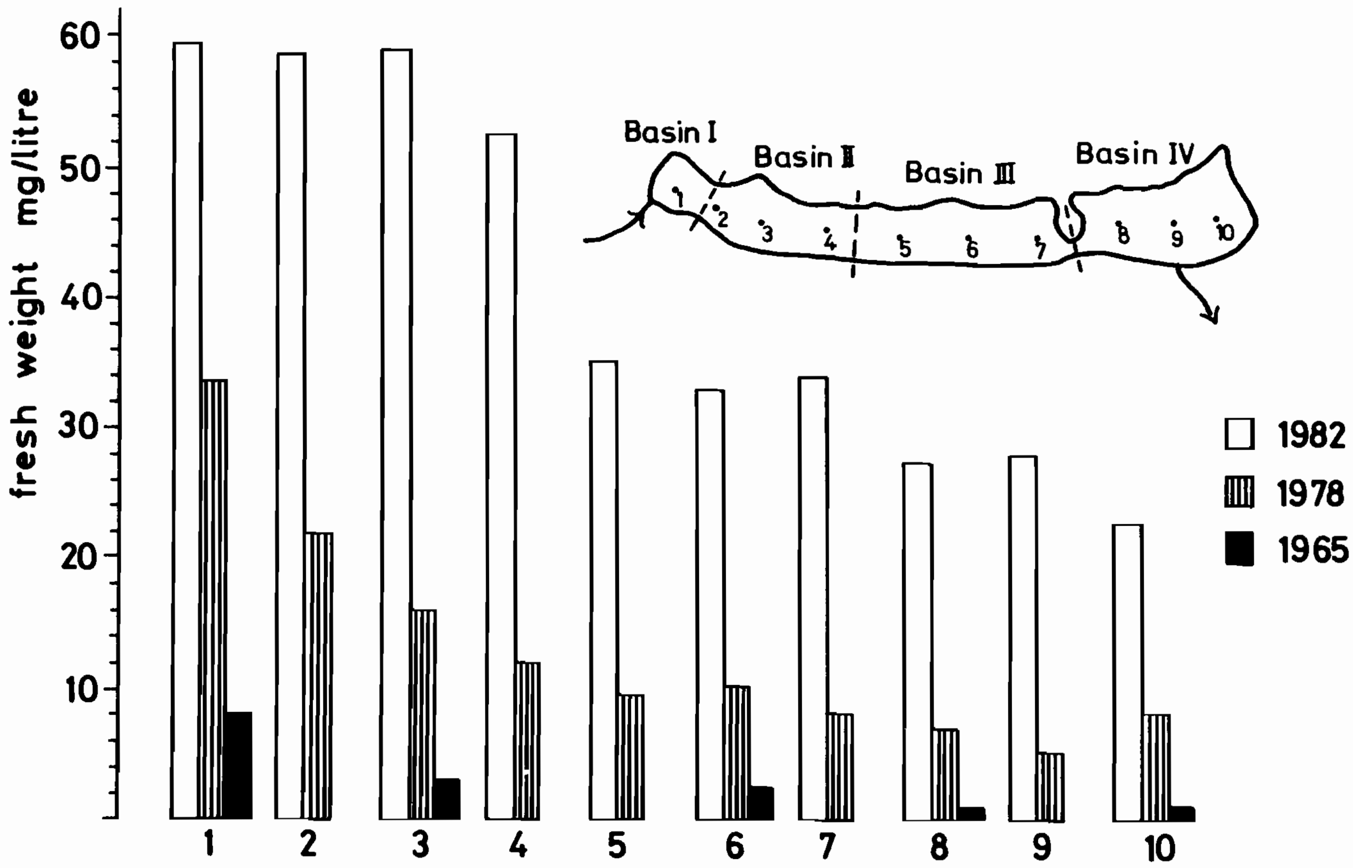
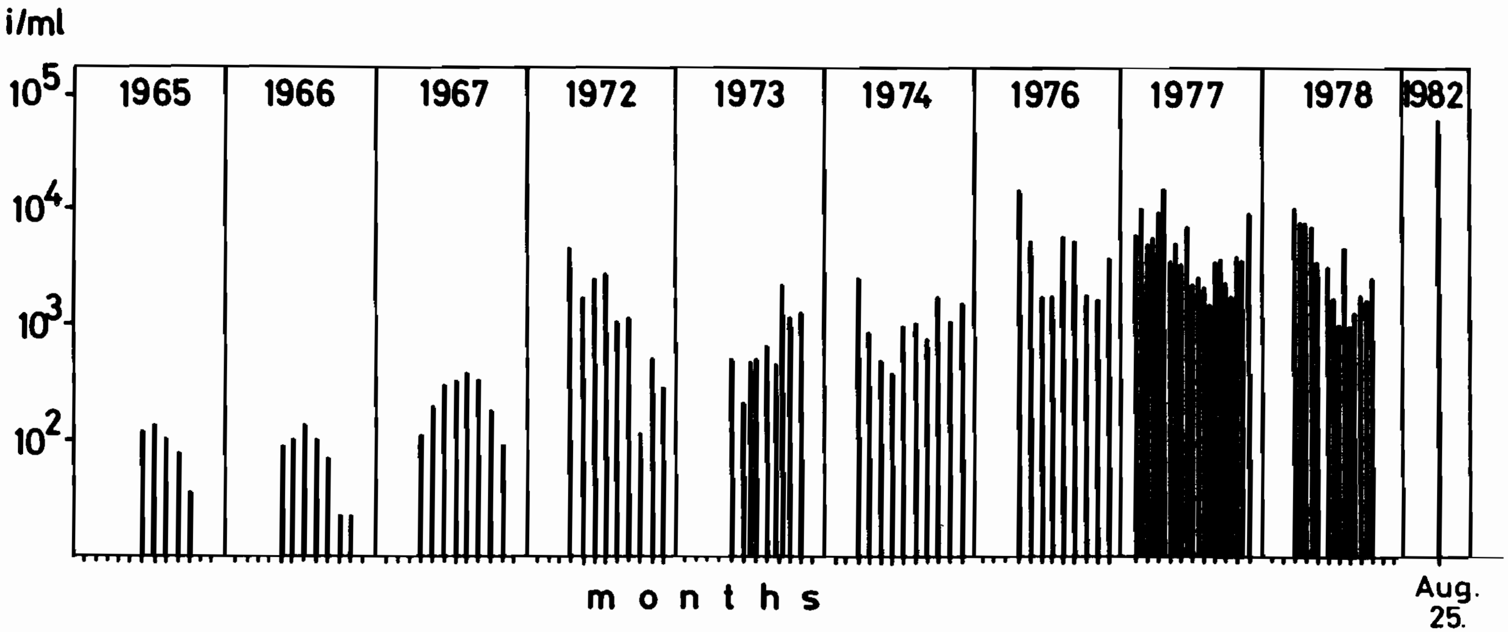


Figure 3. The individual number of phytoplankton in Basin IV /Vörös 1982/

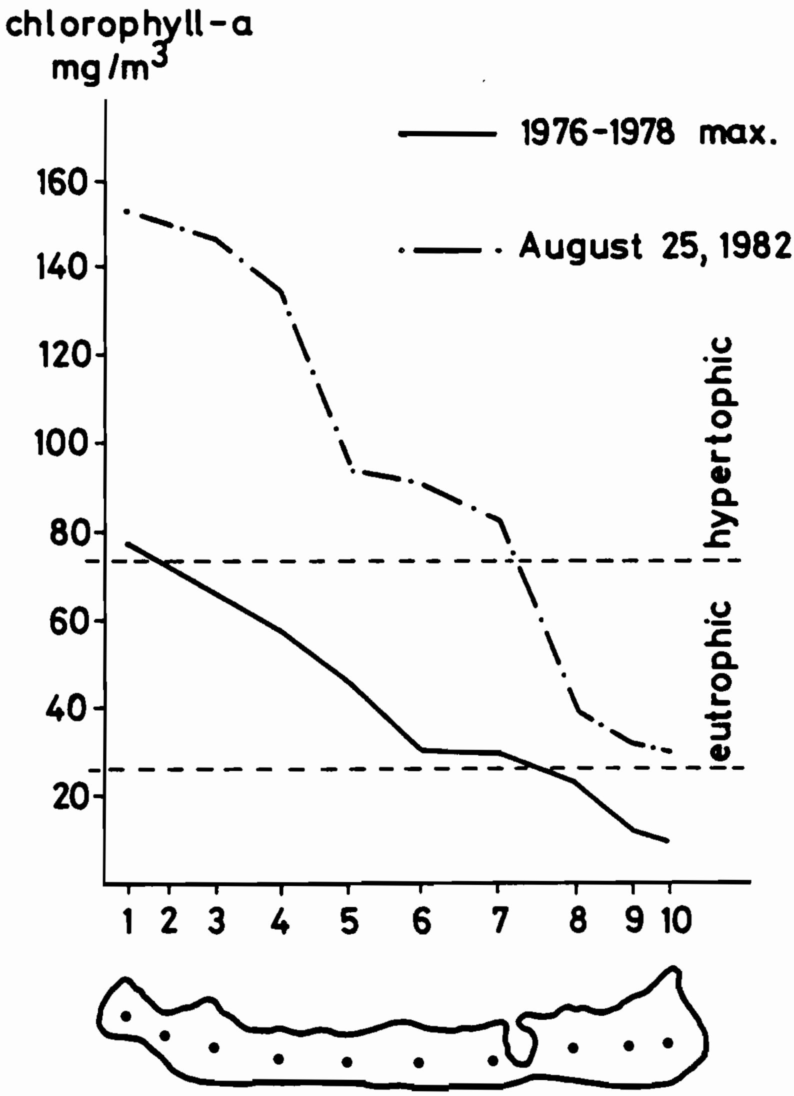


Balaton the biomass and the chlorophyll content were determined parallel with the biomass at Szemes in 1976--1977, at Tihany in 1977 and at Keszthely in 1978-1979 /Vörös 1983/. The correlation between chlorophyll content and biomass was strong in all the three basins. The mean a-chlorophyll content in per cent of the fresh weight were 0.30 at Tihany, 0.36 at Szemes and 0.42 at Keszthely. The extremes were 0.09 and 0.68 per cent.

The chlorophyll data are really numerous. Since 1971 the VITUKI measures the chlorophyll content in 9 stations along the longitudinal axis of the lake and the Transdanubian Water Authority in 16 offshore and several inshore stations at monthly intervals. The results from the two laboratories are in good agreement. Both of them show increasing values toward the Zala River, and in the last years a significant increase was detected at all stations.

The OECD synthesis Report on Eutrophication Control /Vollenweider and Kerekes 1980/ uses a classification, based on the annual maximum chlorophyll content. Between 25 and 75 mg/m^3 lakes are regarded as eutrophic, above this level they are hypertrophic. If we select the maximal chlorophyll values from the data of the period 1976-1978, it turns out, that in this time only the first basin fell into the eutrophic category, the second and third basins showed eutrophic values and the fourth one could be still

Figure 4. The chlorophyll-a concentration in August 25, 1982 and its maxima in 1976-1978



regarded as mesotrophic /Fig. 4/. During the water-bloom in 1982 the a-chlorophyll content was two-three times higher than the maximal values in the 1976-1978 period, the entire lake became hypertrophic, except the fourth basin, but even that showed eutrophic values.

The primary production is one of the best characteristics of the trophic state /Rodhe 1969/.

The first primary production measurements using the C-14 method were carried out in 1962-1963. At that time there were no differences in the productivities of the four basins /Böszörményi et al. 1962/, according to our present terms they corresponded to the mesotrophic level. A more detailed study started in 1972. The production was measured by the C-14 technique fortnightly in four depths in 1972-1973 in Basin IV, in 1973-1974 in Basin I, in 1974-1975 in Basin II, in 1976-1977 in Basin III, and in 1977 again in Basin IV./Herodek and Tamás 1975a, 1975b, 1978, Herodek et al. 1982/.

In Basin IV the production in 1972 was not much higher than in 1962. The maximal daily production was 0.6 g C/m^2. The vertical profile of the production changed according to the actual weather conditions, as the transparency of this shallow water is basically determined by the amount of the sediment swirled up by the waves. In long calms therefore the highest production was measured in the deepest sample, in strong storms on the other hand the euphotic layer was restricted to the uppermost 1 m. Typically, there was photoinhibition in the surface sample, the maximum was at 1 or 2 m depths, and at 3 m photosynthesis was reduced by the insufficient illumination.

In Basin I in 1973 a completely changed figure was found. Due to the self-shading of the phytoplankton here the maximum was even in calm days in the surface, down to 1 m the photosynthesis dropped to the half of the surface value, and below 2 m there was practically no photosynthesis at all. The maximal daily production, 13.6 g C/m^2 belongs to the highest values measured in European lakes. Earlier it was suggested that the high calcium hydrocarbonate content of the water will prevent eutrophication by precipiting the phosphate. This extremely high production however proved racing eutrophication, and on this basis we gave the warning in 1973, that if the nutrient load will not be reduced the water quality of the whole lake will be seriuously damaged within a decade. Unfortunately this prediction proved to be right.

In Basin II the vertical profiles of the photosynthesis were similar to those of Basin I. The production was 3-4 times higher than in 1963. In July 1974 a maximal daily production of 3.1 g C/m^2 was found. This year the autumn was very rainy, and increased the diffuse loading, which can be an important nutrient source especially in this basin. Probably this extra load caused the extremely strong water colouration by diatoms which lasted from October to February. That year the lake was not frozen and the production remained high during the whole winter. In February a strong fish kill occurred in this basin. Its origin was not unequivocally clarified, but it is possible that the anaerobic conditions in the deeper water layers contributed to it.

In Basin III /1976-1977/ the vertical profiles of the primary production depended still on the wind velocity i.e. the algae did not alter the original optical properties of the water, but the production was much higher than in Basin IV, and attained the 2.6 g C/m^2/day. The primary production shows a definite seasonality. It has a smaller peak during the spring diatom outburst, diminishes in May, and increases again from the middle of June. With increasing eutrophication the summer peak became more expressed.

In 1977 the measurements showed twice as high primary production in Basin IV, than in 1972, and the maximal daily production was 1.7 g C/m^2, proving that in this year the whole lake was already in the state of racing eutrophication, including this less eutrophicated basin.

The annual productions as calculated from these measurements were the followings:

Basin I /32 km^2/: 830, Basin II /120 km^2/: 301, Basin III /216 km^2/: 274, Basin IV /228 km^2/: 182 g C/m^2 /1977/.

This means, that about 26 560, 36 120, 39 312 and 39 672 metric tons of carbon were photosynthetically fixed in the four successive basins, and the whole production of the phytoplankton could be estimated to about 141 664 metric tons/year in the mid-seventies.

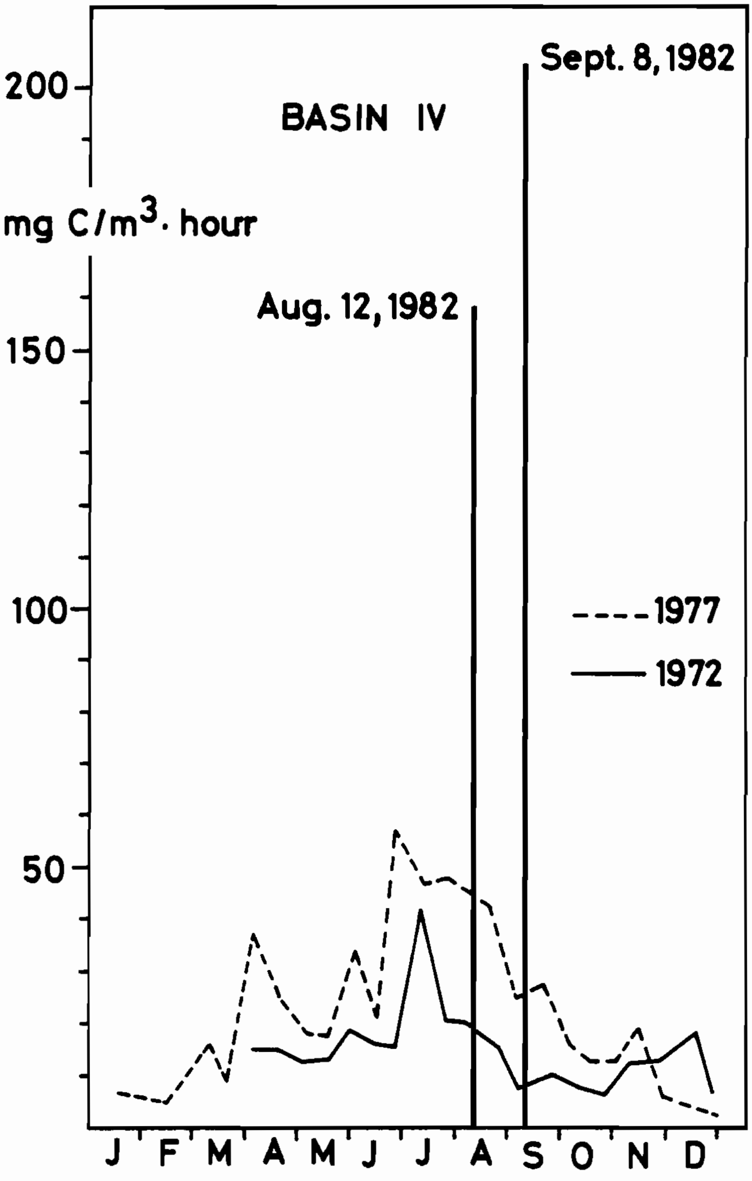
In June-October 1979 the measurements were repeated in Basin I /Vörös et al. 1983/. The production in the optimally illuminated layer was similar to that in 1973, but that related to the surface area diminished already. This is a consequence of the increased phytoplankton biomass, that restricted the euphotic layer to half metre. This demonstrates that in a nutrient rich water, where the primary production is basically light limited, by progressing eutrophication the production may decrease. The production was measured simultaneously with both the C-14 and O_2 techniques, and the two methods gave good parallels.

The field measurements are time consuming and due to the distances it seems difficult to work in all basins simultaneously. In 1978 therefore water samples collected in 10 stations along the longitudinal axis of the lake were brought to the laboratory where the labeled hydrocarbonate was added, and the samples were incubated in a photoluminostate for four hours at the ambient water temperature. In this case the illumination was fixed at 4000 lux, therefore the production per surface area can not be calculated, but useful results were obtained for the relative productivities. In all cases the production showed a regular gradient increasing toward the Basin I. The increase in south-west direction is demonstrabile also within the basins. The ratios of the productivities of the basins found in these laboratory experiments are in good agreement with those, measured in the field.

Systematic field measurements will start again in 1983. The water bloom in 1982 prompted however some measurements in Basin III and Basin IV. At both stations the euphotic layer became restricted to the upper 2 m, as below that level the illumination was less than 1 per cent of that of the surface. In the optimally illuminated layer in Basin III five times in Basin IV /Fig. 5/, four times higher productions were measured than the highest values found in 1976 and 1977.

In these studies primary production, phytoplankton biomass and light intensity were simultaneously measured in four depths. The light optima of the phytoplankton could be estimated from the several hundred data obtained using

Figure 5. The primary production in the optimally illuminated water layer in Basin IV

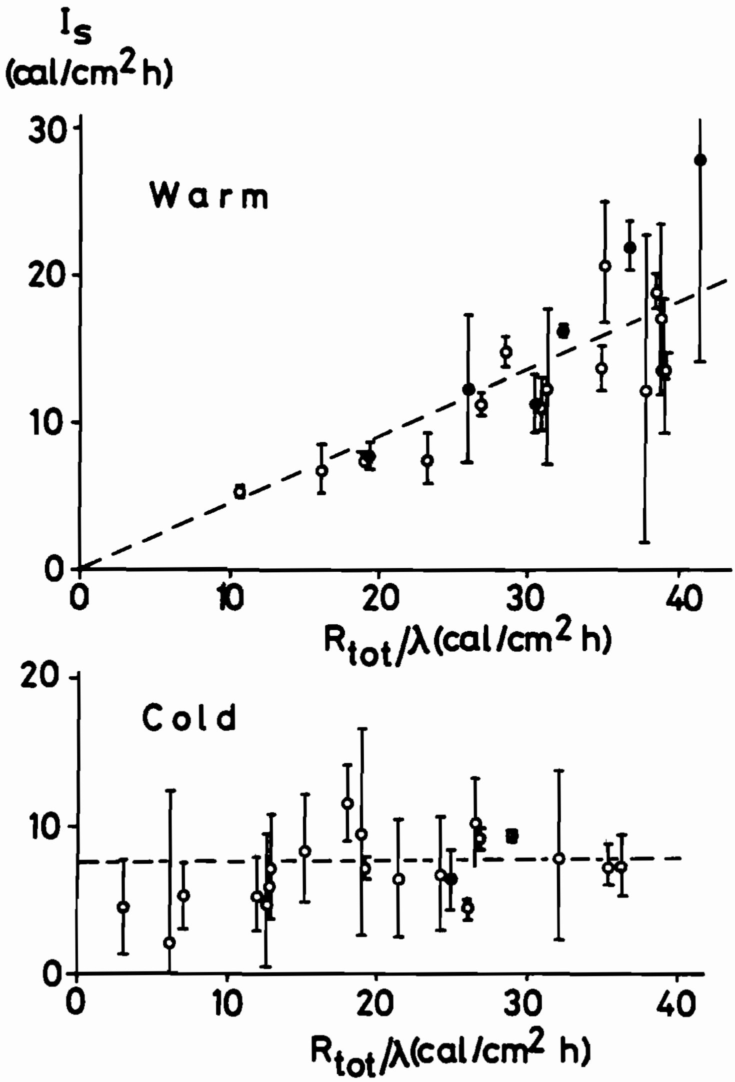


a nonlinear minimum squares method /van Straten and Herodek 1982/. In the cold water communities the light optimum was 8 cal $\cdot$ cm^{-2} $\cdot$ $hour^{-1}$. In the summer phytoplankton it ranged from 10 to 20 cal $\cdot$ cm^{-2} $\cdot$ $hour^{-1}$, and the actual value of the light optimum increased linearly with the illumination /Fig. 6/, thus in this period there is a definite light adaptation. The cold water phytoplankton showed the fastest growth at 8°C. In summer the highest daily production/biomass ratios were found at 24-26°C. This ratio /if both production and biomass are expressed in carbon/ attained 4.7 in the optimally illuminated layer in Basin III, but in Basin I values above 10 were also found, i.e. in Lake Balaton the turnover of phytoplankton biomass is very rapid.

The limiting nutrient problem

It is generally supposed, that lakes in the temperate zone are originally phosphorus limited, because the ratio of phosphorus to other nutrients is much less in the water than in the living cells. In the sewage waters on the other hand the N/P ratio is lower than in the organic matter, mainly due to the high phosphorus content of detergents. Therefore polluted lakes can turn from phosphorus

Figure 6. The saturation light intensities as a function of total average irradiation on the experimental day /points for Basin I distinguished by closed circles/ /Van Straten and Herodek 1982/



I_s = saturation light intensity,
R_{tot} = total irradiance, λ= daylength

to nitrogen limitation. In such waters then the nitrogen fixing blue-green algae have a special adventage, and become dominant in the plankton. Beyond this reasonable simplification there remain however many theoretical and methodical problems concerning the limiting factors. It should be realized, that the term "limiting nutrient" is used at least in two different senses. One is the factor that limits the momentary growth of the plankton, and the other is the nutrient that determines the maximal biomass attained during the year. At the present there is no absolute method to determine either of these factors, but with a combination of different techniques the problem can be approached.

The simplest method is the chemical determination of the N/P ratio in the water. It is supposed, that above 15 the water is phosphorus limited, below 7 it is nitrogen limited, and between these two values there is double limitation. According to the chemical data of the VITUKI, stored in the IIASA Balaton files, in the period of 1976-1978 the average total N/total P ratio was about 20, suggesting frequent phosphorus limitation, but large seasonal variations occurred. In 1981 this ratio was in most cases lower than 15, and in the Basin I values below 5 were also found /Istvánovics 1982/. The dissolved inorganic nitrogen compounds and the orthophosphate are certainly utilized by algae, but at the present chemical analysis can not tell us, what parts of the other phosphorus and nitrogen forms are biologically available, therefore the N/P ratios contain important informations, but they do not solve the problem unequivocally.

In the algal bottle tests the lake water, inoculated with a test alga, most frequently with Selenastrum capricornutum, is enriched with different nutrients. The nutrient, increasing most the biomass during the test is regarded as the limiting factor in the original lake water /Miller et al. 1978/. Dobolyi and Ördög /1981/ made such experiments with water samples collected in Basin I and Basin IV. They found phosphorus limitation, but the authors themselves refrained from generalisations on the basis of the few measurements. It is a special problem at the interpretation of these experiments, that the water is filtered prior to the biotest to remove the original phyto- and bacterioplankton, but this filtration retains all other particulated nutrients too. Availability during such tests may differ from that under natural conditions, due to the absence of bacteria. Despite of these inherent problems of the method, it seems worth to continue the assays with more frequently sampled lake water.

The measurement of primary production in water samples enriched with different nutrients /Goldman 1960/ became a widely used technique to study the nutrient limitation of the natural phytoplankton. It seemed to be an ideal method, revealing the immediate reaction of the natural phytoplankton. Unfortunately during our preliminary experiments we obtained no immediate increase in carbon fixation, but in accordance with the experiments of Lean and Pick /1981/ the addition of orthophosphate even decreased the production for a period. It seems, that the phosphorus defficient cells used their energy first for the phosphate uptake, and the photosynthetic carbon fixation increased only after the phosphorus was depleted from the water, or when it attained already a certain level within the cells. After the addition of the nutrients the water samples must be incubated therefore for a few days before the measurement of carbon fixation. This way even this method can not measure the momentary limitation, however it maintains the advantage that it works with natural plankton.

The limitation was studied with this procedure in Basin I and Basin IV from August 1980 to August 1981 /Istvánovics 1982/. Water samples of 200 ml were enriched with 20 µg/litre PO_4-P, with 140 µg/litre NO_3-N, and with both nutrients.

These samples and the unenriched controls were then incubated by ambient lake water temperature and by 4000 lux illumination. After four days $Na_2{}^{14}CO_3$ was given into the flasks and they were further incubated for 3 hours. The samples were then filtered and from the radioactivity of the algae and the total carbonic acid content of the water the carbon fixation was determined.

In samples collected in Basin IV the production was increased in the early spring by phosphorus, in the late spring and in summer by nitrogen, in autumn again by phosphorus and in winter by none of the nutrients.

In Basin I there was no such regular seasonality. Both in spring and summer the phosphorus and nitrogen limitations alternated. In addition, while in Basin IV the production was increased by only one of the two nutrients at one time, in Basin I both phosphorus alone and nitrogen alone could increase the production in the same sample. In Basin IV the two nutrients given together increased much more the carbon uptake, than the limiting nutrient by oneself, while in Basin I this difference was not so expressed.

In Basin I there are already blue-greens, which could fix nitrogen from the atmosphere, therefore their growth is phosphorus limited. Besides the blue-greens there are at the same time also other algae, unable to nitrogen fixation, they are therefore nitrogen limited. This can explain why both nutrients alone increased carbon fixation in samples from Basin I. In Basin IV there is usually high nitrate concentration in early spring, and the phytoplankton is correspondingly phosphorus limited. The nitrate is exhausted from the water during the diatom outburst, with results in nitrogen limitation in late spring. The nitrogen limitation in summer of 1981 showed that already this part of the lake is also overloaded with phosphorus and in 1982 we had indeed the blue-green invasion also here.

The phosphorus metabolism

In water chemistry orthophosphate is determined by the reaction of orthophosphate with molybdic acid /Murphy and Riley 1962/. Other phosphorus compounds must be first digested to release ortophosphate. In a routine analysis from one part of the water samples the total phosphorus content is determined after acidic digestion, the other part is filtered through membrane filters of 0.45 µm pore size. From one aliquot of the filtrate the dissolved reactive phosphorus is determined without previous digestion, and is usually regarded to represent the original orthophosphate content of the water. Another aliquot is digested prior to the reaction with molybdic acid, and it gives the amount of the total dissolved phosphate. Particulated phosphorus can be calculated as the difference between the total phosphorus and the total dissolved phosphorus.

These phosphorus fractions have been determined from monthly samples of the nine stations by L. Tóth from 1971 on, and the unpublished data are stored on the IIASA files. Along the longitudinal axis of the lake the total phosphorus concentration changes according to a gradient similar to those of the phytoplankton biomass and production, but the slope of the phosphorus curve is somewhat lower. In 1976-1978 the mean concentrations of total phosphorus were 78, 54, 38 and 29 µg/litre in the four successive basins. The total dissolved phosphorus amounts to about the half of the total phosphorus. The dissolved reactive phosphorus level is low, the mean values varied between 4 and 6 µg/litre in the different basins.

Unreactive dissolved phosphorus is generally believed to represent dissolved organic phosphorus compounds. On the contrary Dobolyi /1980/ found that after UV treatment of the filtered Lake Balaton water a large part of the dissolved phosphorus remained unreactive. The excessive UV treatment oxidized all organic compounds, therefore the unreactive phosphorus must be present in inorganic form, i.e. in condensed phosphates. In the central basin of the lake these condensed phosphates amounted to 2-67 µg/litre, except in winter, when under the ice the concentration was only 0.2 µg/litre. In the next step, the condensed phosphates were idenfified by gel chromatography. First bivalent cations were removed by passing the water through a cation exchanger resin to prevent lime precipitation during the subsequent concentration of the samples which could remove some phosphorus too. Here upon the water was treated by UV to oxidize all organic compounds, and the solution was concentrated by freezing-drying. The chromatography of such lake water samples on Sephadex-25 fine columns revealed phosphorus compounds up to a condensation degree of 10. The condensed phosphates of the Zala river, of the effluent of a sewage treatment plant and of a photooxidized alga suspension were analysed with the same method. The elution curve of the lake water sample resembled that of the photo-oxidized algal suspension /Fig. 7/, suggesting that the polyphosphates in the lake are produced in the phytoplankton. Algae are known to contain significant polyphosphate reserves /Rhee 1973/, and it seems possible that during the degradation of the cells these are released into the water. It is known that algae and especially bacteria can utilize the polyphosphates dissolved in the water, but we do not know the speed of such processes under natural conditions. A better understanding

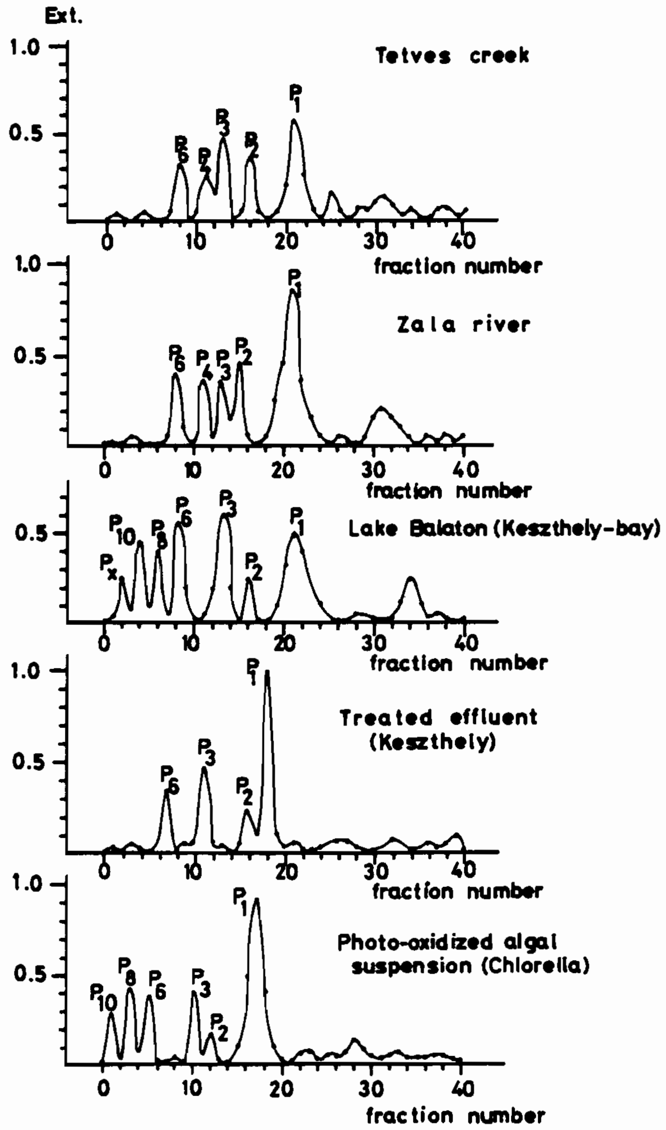


Figure 7. The condensed phosphates in the Tetves creek, in the Zala river, in Lake Balaton, in a treated sewage and in a photooxidized algal suspension /Dobolyi 1980/

of the role of polyphosphates awaits therefore for further kinetic studies.

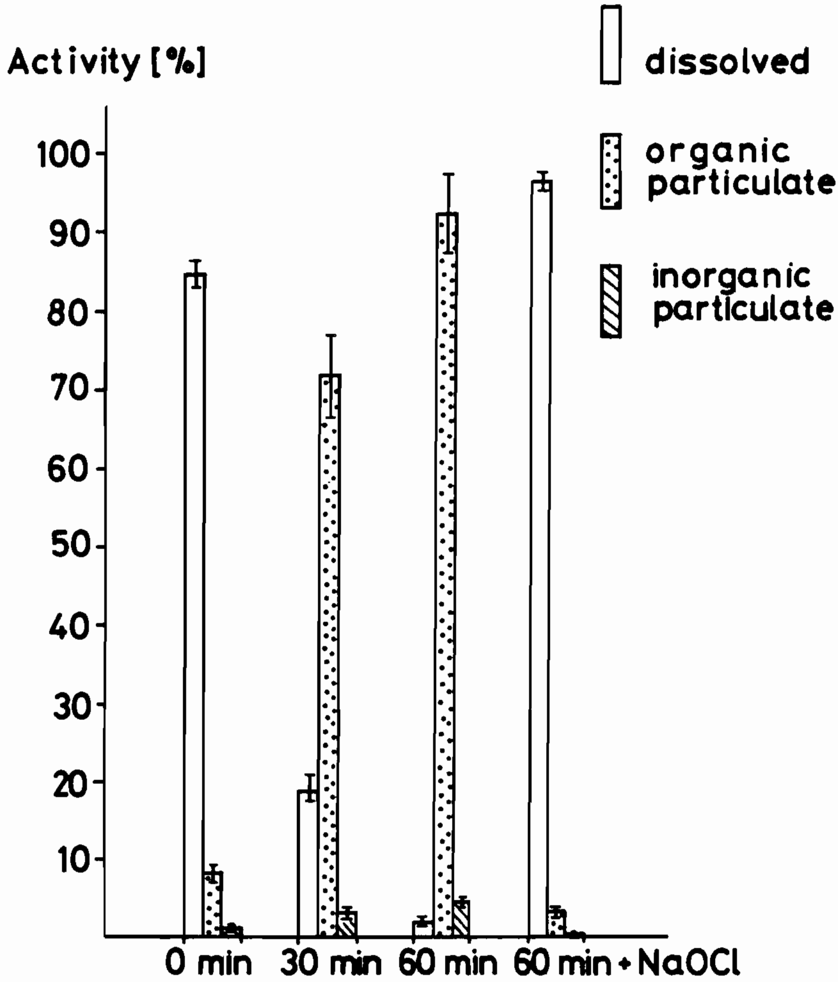
The most important step in the phosphorus cycle in lakes is the orthophosphate uptake by the algae. This process was studied both in an enclosure in the lake and in vitro in the laboratory. In this shallow lake during heavy storms the amount of suspended solids can reach 200 mg/litre. Biogenic lime precipitation is also a major process in this water, where the main ions are Ca^{2+} /32 mg/litre/ and HCO_3^- /280 mg/litre/.

It was therefore suggested by Oláh et al. /1977/ that the ortophosphate level in the lake is primarily controlled by the adsorption to the suspended sediment or to the freshly formed biogenic lime. On the contrary our experiments /Dobolyi and Herodek 1980/ show, that it is the active uptake by algae that determines the orthophosphate level also in this lake.

First a plexi glass box, open at the bottom was placed in the shallow water, the water within this enclosure being stirred by an electrically driven paddle. By phosphate addition the PO_4-P concentration was raised from the original 2.4 µg/litre to 25 µg/litre, and the dissolved reactive phosphate concentration was measured at different times. In one hour it dropped down to the original 2.5 µg PO_4-P/litre. On the contrary, if the algae were killed by NaOCl, the PO_4-P concentration remained steadily 25 µg/litre. This proves that the phosphorus elimination is connected with life processes. It remained however possible that the phosphorus was removed by biogenic lime precipitation, induced by algal photosynthesis. In the third experiment therefore EDTA-Na_2 was also added to the water in the box to maintain the Ca^{2+} in complex. It was found, that even in this experiment, where the biogenic lime formation was prevented, the added phosphate disappeared at the same speed, as in the parallel without EDTA-Na_2 addition. It seems therefore that neither suspended sediment, nor biogenic lime, but the algae themselves take up primarily the phosphate from the water.

The same conclusion was drawn from the in vitro experiments. Carrier free $H_3{}^{32}PO_4$ was added to the water samples, and after 30 and 60 min the water was filtered through membrane filters of 0.45 µm pore size. The radioactivity of the filtrate was directly measured and regarded to represent the orthophosphate remaining in solution after the incubation. The membrane filters with the retained particulate material were photooxidized. The radioactivity, that remained in the particulate form after this UV treatment represents the phosphorus incorporated into inorganic material, while the fraction that went into solution corresponds to the phosphorus utilized by algae. In 30 minutes 72 per cent of the added labeled orthophosphate was incorporated into the particulated organic fraction, and only 3 per cent into the particulated inorganic matter /Fig.8/.

Figure 8. The distribution of radioactivity among the different phosphorus forms after in vitro incubation of lake water with $H_3{}^{32}PO_4$ /Dobolyi and Herodek 1980/



The turnover time of the orthophosphate in lake water was studied by isotope technique in water samples collected in Basin I and Basin IV in 1980-1981 /Istvánovics 1982/. Carrier free $H_3{}^{32}PO_4$ was added to water samples of 200 ml, and at different times, usually after 2, 4, 8, 16 and 32 minutes subsamples of 5 ml were passed through membrane filters and the radioactivity in the filtrate was measured. The uptake showed first order kinetics. The turnover times proved to be very short in both basins and changed seasonally. Under the ice they were relatively long, 100 min in Basin I, and 401 min in Basin IV, but in early spring values below 10 min were found at both stations. In late spring there was a second maximum, 335 min in Basin I and 97 min in Basin IV. In the summer the turnover times were 1-4 min in Basin I, and 4-8 minutes in Basin IV, while in October they rised again. The uptake rate equals the concentration of orthophosphate in the water divided by the turnover time. If we divide the concentrations of the soluble reactive phosphate by these turnover times, unrealistically high uptake rates are obtained. The algae would take up more phosphorus than carbon, which is not very likely. It was however demonstrated by Rigler /1966/ that the generally used molybdate technique overestimates the orthophosphate, especially in low concentrations. Rigler added different amounts of unlabeled phosphoric acid together with the carrier free labeled phosphoric acid to lake water samples, measured the turnover time of the phosphate, and calculated the uptake velocity as the sum of the original and added phosphate per the turnover time. The curves showed decreasing uptake by increasing concentrations, which contradicts to all biological expectations. He got the Michaelis-Menten type curves only by supposing that the original phosphate concentration in the water is much lower than the one measured by the molybdate technique. This method can not tell us the real values, but it can prove that the measured ones are overestimates. Similar experiments with Lake Balaton water showed that in most cases the real concentration must be much lower than the chemically determined one. To evaluate these results the turnover times were plotted as a function of the amount of added phosphate.

According to the Michaelis-Menten formula

$$v = V \frac{S+A}{K+S+A}$$

where v = the uptake velocity

V = the uptake velocity at substrate saturation

S = the original substrate concentration

A = the concentration of the added substrate

K = the half saturation constant.

As the uptake velocity $v = \frac{S+A}{T}$

where T is the turnover time of the substrate in the water,

$$T = \frac{A}{V} + \frac{K+S}{V}$$

The straight lines obtained proved that the phosphate uptake followed really the Michaelis-Menten kinetics. From such graphs /Fig. 9/ we can determine the sum of S and A and the value of V max. Even by this technique we can not obtain the original orthophosphate concentration, but it must be lower than the S+K value. In ten from the fifteen occasions studied, S+K proved to be lower than the S measured by molybdate.

The S+K values are higher in the hypertrophic Basin I than in Basin IV. It is very likely that S and K are in positive correlation due to the adaptation of algae to the external nutrient concentrations, and in this case the orthophosphate level is usually higher in Basin I than in Basin IV.

The maximal uptake velocities, that would be attained at nutrient saturation are also much higher in Basin I than in Basin IV. These values are the highest in late summer during the blooms of the blue-greens. In this period the biomass is also maximal, and the nitrogen fixation renders the system phosphorus defficient.

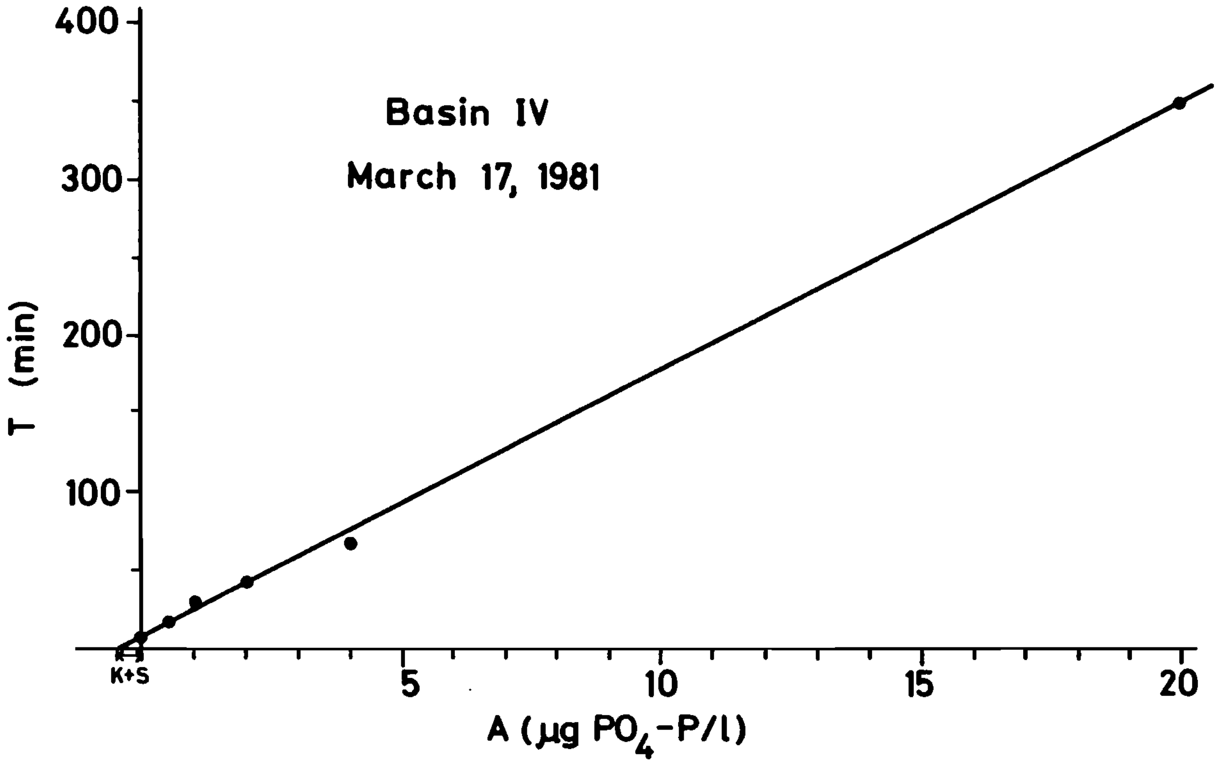


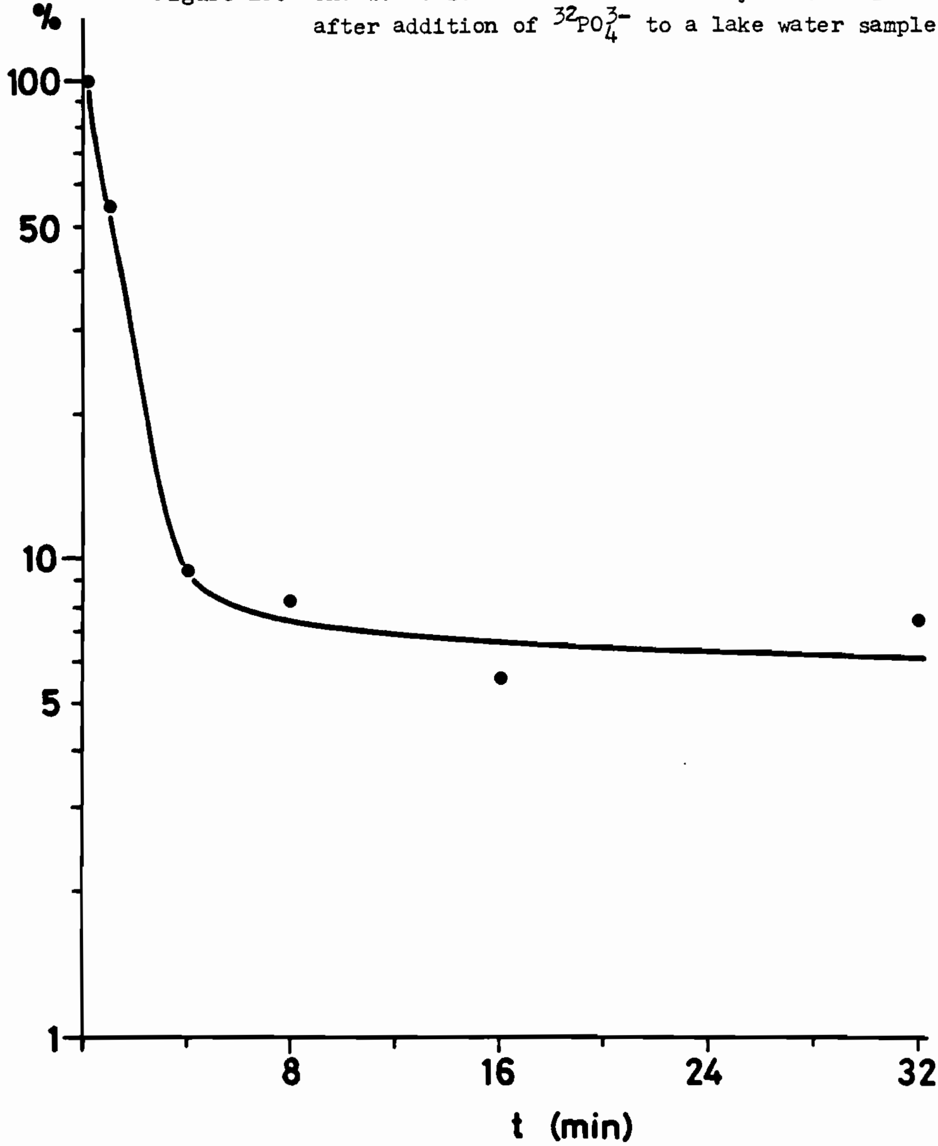
Figure 9. The turnover time of the orthophosphate in lake water samples as a function of the added PO_4-P

The turnover time was similar in the two basins of different trophic states, because in the first basin both S+K and V are higher than in the last one, and $T = \frac{S+K}{V}$.

The determination of S+A and V is therefore more informative than only the measurement of the turnover time.

It would be of course wellcome to determine both the phosphate content of the phytoplankton and the orthophosphate level in the water. At the present, there are no techniques to separate algae from other suspended solids, and as we have seen, the chemical measurement overestimates the real orthophosphate concentration, mainly due to the hydrolysis of other phosphorus compounds at the low pH of the molybdate method. From the kinetic studies however we can get some idea about the relation of these two compartments. If the log of the radioactivity in the liquid phase is plotted as a function of time we get first staight lines, but the slope of these gradually decreases, and finally they run parallel with the time axis /Fig. 10/. It seems therefore, that a steady state is attained after a longer time, when 1-10 per cent of the original activity is in the soluble phase, while the rest in the particulate form. If we consider this situation as the result of a dynamic equilibrium between the algae and the water, then in the water there is 10 to 100 times less orthophosphate than the phosphate content of the algae. We know also,

Figure 10. The decrease of the radioactivity in solution after addition of $^{32}PO_4^{3-}$ to a lake water sample



that in algae there must be different phosphorus pools like nucleic acids, phospholipids, phosphoproteids, sugar phosphates, nucleotids, polyphosphates etc. The compartment equilibrating with the water is therefore smaller, than the total phosphate content of the phytoplankton, and the two compartment hypothesis is also of limited validity. According to the literature, the phosphorus content in algal cells used to vary from 0.2 to 1 per cent of dry weight. It seems therefore likely that in Lake Balaton and especially in Basin IV the PO_4-P level in the water is usually below 1 µg/litre.

The nitrogen metabolism

Parallel with phosphorus also the nitrogen forms were analysed by L. Tóth in samples collected monthly at the nine stations along the longitudinal axis of the lake from 1975 on, and the unpublished data are stored on the IIASA Balaton files. The nitrogen forms were studied in the course of the nitrogen fixation /Oláh et al. 1981/ and the limiting nutrient /Istvánovics 1982/ studies, too.

In 1976–1978 the mean concentrations of total nitrogen were 1.4, 1.2, 0.9 and 0.7 mg/litre in the four successive basins. The total dissolved nitrogen amounts to about two thirds of the total nitrogen. The NH_4-N concentration in Basin I sometimes attains the 50 µg/litre, and shows strong seasonal and diurnal changes. In Basin IV it is around the limits of detectability. The nitrite concentration is only a few µg/litre. Among the inorganic nitrogen compounds the nitrate has the highest concentrations, but it varies seasonally. The highest values were measured in early spring after ice break, at this time the NO_3-N concentration reaches several hundred µg/litre. This reserve is then utilized by the diatom outburst. In summer the nitrate concentration remains low, but in autumn increases again. For the ammonia and nitrate uptake of the phytoplankton we have no data at the present, experiments with ^{15}N labeled compounds are starting only now.

The nitrogen fixation on the other hand was studied in details from 1977 to 1980 /Oláh et al. 1981/ with the acetylene reduction technique. This method, introduced into the limnology by Stewart et al. /1967/ is based on the ability of the nitrogenase enzyme, responsible for nitrogen fixation to reduce acetylene to ethylene. This method is more sensitive, and above all much cheaper and less time consuming than the direct measurement with $^{15}N_2$, therefore it became widely used.

Acetylene was injected into bottles containing 150 ml lake water, and after thorough shaking the bottles were suspended in the lake at the same depths, from where the water was collected. After the 3 hours of in situ exposal the bottles were again vigorously shaken, and a gas sample was taken for analysis. The Pye Unicam GCV gas chromatograph worked with a glass column of 2.1 m length and 4 mm diameter, filled with Poropack-N. The temperature was 80 C^{o}, the carrier gas was N_2 with a flow rate of 40 ml/min. A flame ionization detector was used.

The nitrogen fixation was very high in Basin I, in Basin II it was by one, in Basins III and IV by two orders of magnitude lower, which corresponded well to the distribution of the blue-green algae at the time of the experiments. For in summer of 1982 the blue-greens became abundant in all parts of the lake, it is very probable that the horizontal distribution of the nitrogen fixation changed basically.

The seasonality of the nitrogen fixation corresponds equally well to the occurrence of blue-greens. It is low in June, increases in July, it has the maximum in August when the blue-green population is in the log phase, in September it is still high, then it drops rapidly parallel with the destruction of the blue-green algae.

The nitrogen fixation was usually the highest in the forenoon, but sometimes a second peak appeared in the late afternoon. The activity during night was also appreciable, i.e. the algae can use besides the light energy also that of dark respiration.

In the well illuminated layer the nitrogen fixation was usually higher than near the bottom, and these vertical differences were more expressed in August-September i.e. in the time of the maximal activity than in June.

From the measurements between 1977 and 1980 the nitrogen fixation in the warm period in Basin I is estimated to 7-13 g N/m^2. The annual fixation must be somewhat higher.

The nitrogen fixation was measured also in the sediment /Oláh et al. 1981/. Intact cores were taken into tubes of 5 cm diameter, and through the perforations on the wall of the tube acetylene was injected into the sediment. After incubation the sediment was shaken, the gases collected and analysed like in the case of the experiments with water. According to these studies the annual nitrogen fixation in the sediment of Basin I is about 4 g $N/m^2/yr$, which is higher than in most other lakes. In the sediment the nitrogen is fixed by bacteria.

The Zala river transports about 1000-1500 tons/yr of nitrogen into Basin I /Joó 1980/. This corresponds to an areal loading of about 35 g $N/m^2/yr$. It seems therefore that on an annual basis two thirds of the total N input of Basin I are of external origin, but in the critical summer period the nitrogen fixation is the main nitrogen source.

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HYDRODYNAMIC AND MASS TRANSPORT ASPECTS OF THE LAKE BALATON MODELS

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INTRODUCTION

An important part of the Lake Balaton Case Study has been the study of circulation in the lake and the development and comparison of mathematical models with which to simulate circulation. The information and models developed have been used to further the understanding of mass transport as an influence on lake water quality and eutrophication. This general report discusses the character of circulation in shallow lakes, the prediction of wind-driven circulation using mathematical models, and the role of circulation in establishing lake water quality and trophic status. The report draws upon results from the Lake Balaton Case Study for specific examples and presents the major findings achieved with the Balaton circulation models. The Balaton circulation models include one, two and three-dimensional versions, thus permitting a comparison of model formulations for shallow lakes. As well, the study included detailed study of the linkage of circulation model results with water quality models, and the results of those investigations will also be summarized in this report. The report closes with an examination of the major findings of the Lake Balaton studies to gain perspective on the state-of-the-art in modeling shallow lake circulation and to indicate specific areas where future research would be beneficial.

CIRCULATION MODELING OF SHALLOW LAKES

Shallow Lake Dynamics

For the purposes of circulation modeling, a shallow lake is one without density stratification. Although Entz (1976) has observed occasional weak temperature stratifications in Lake Balaton, the density gradients are neither sufficiently persistent nor steep to significantly effect the circulation dynamics of the lake. Lake Balaton is thus a large and very shallow lake.

There are two major classes of circulation induced in a shallow lake. The first is the hydrologic through-flow established by the net inflow and outflow to the lake. Although the velocities associated with this circulation are often extremely small, the motion is important as a long term influence on water quality. This is because hydrologic flow alone accounts for the net import and export of nutrients to the lake.

The second major class of motion is wind-induced circulation. When the wind blows across the lake surface, it exerts a shear upon the water to produce a set-up. Set-up is the phenomenon wherein the water surface at the downwind side of the lake is superelevated by the wind force above the elevation of the upwind side. Set-up is enhanced in shallow lakes as compared to deep lakes. Cessation of the wind removes the force upon the surface and releases the set-up water. The resulting motion produces a seiche: the wave-like flow of water back and forth within the lake.

Strong seiche motion has been observed in Lake Balaton by Muszkalay (1973). Muszkalay collected wind, current and water surface elevation data on the lake during a period of approximately ten years. He employed his excellent observation data base to develop empirical formulae relating set-up and current velocities to wind speed and duration. These formulae serve as the standard for calibration of the circulation models described below.

The force of the wind upon the lake leads to other manifestations as well. Based upon steady-state hydrodynamic analysis, the vertical velocity profile in a shallow lake may be predicted. The profile is characterized by a surface current aligned with the wind, with an underlying bottom return current directed oppositely. In a large lake, this simple cycle of flow is complicated by transience and nonuniformity in the wind field. Local nonuniformities are particularly conspicuous in Lake Balaton due to modification of the wind field by adjacent hills to the north. As a consequence of the topography, significantly different wind speed and direction are observed simultaneously at different locations along the lake.

Mathematical Models

A variety of alternative lake circulation models are available for application to different types of lake geometries and physics. Two general types of models may be defined based upon their spatial representation: circulation models, which simulate two or three-dimensional flow, and simplified models with fewer dimensions.

Simplified models include zero and one-dimensional models. Zero-dimensional models are rarely classed as circulation models since they do not consider water motion other than a specified through-flow or mass balance. Such models often appear in water quality modeling in lieu of a more detailed accounting or water motion, however. One-dimensional models, though generally not classed as circulation models per se, are able to account for wind and other forces upon the lake and thus better capture the dynamics of internal lake motion. One-dimensional vertical models are in wide use, but for application to deep stratified lakes. One-dimensional longitudinal models are applicable only to long and narrow lakes -- of which Lake Balaton is one example. Below, we report on a one-dimensional model application to Lake Balaton by Somlyódy (1982).

Two and three-dimensional models are generally classified as true circulation models. These models consider the forces of wind and bottom friction, as well as the influence of inflows and outflows, to predict the motion within the lake. A number of major alternatives are available. Longitudinal two-dimensional models are applicable to long, narrow and deep lakes -- typically stratified river impoundments. Horizontal two-dimensional, or single layer, models are widely applied to lakes assumed to be vertically uniform. These models are well suited to shallow lakes, and we report an application by Shanahan and Harleman (1982) to Lake Balaton in the discussion to follow. However, single layer models are unable to predict variation of current velocity over the lake depth. If such vertical information is desired, a three-dimensional model must be used. One alternative is the fully three-dimensional model in which the variation of vertical and horizontal velocity components over all three spatial coordinates is determined. This type of model is extremely expensive to employ and thus rarely used. More effective alternatives are multi-layer models which simulate the lake as a series of vertical layers. Within each layer, motion is assumed to be horizontally two-dimensional. A layer interacts vertically with adjacent layers at its upper and lower boundaries. Two types of multi-layer models exist, each based on a different treatment of the inter-layer conditions. A further simplification of three-dimensional lake motion is made in the Ekman-type models which depend upon a numerical solution of horizontal current structure in concert with an analytical solution for the vertical structure. Ekman-type models have been examined for application to Lake Balaton by Shanahan, Harleman and Somlyódy (1981) and Fisher (1980).

LAKE BALATON CIRCULATION MODEL STUDIES

Early Investigations

In the course of the Lake Balaton Case Study, numerous circulation model alternatives have been investigated. A variety of different model types was examined in the earlier phases of the Balaton Study as practical and useful modeling alternatives were sought (Somlyódy and van Straten, 1980). The earliest of these was a trial of Lick's (1976) Ekman-type model on the IIASA computer. That model was originally developed for application to Lake Erie, but proved to be too inflexible to be simply adapted for application to

Balaton. Another Ekman-type model, by Young and Liggett (1977), was programmed for Lake Balaton and tested by Fisher (1980). Although the model was sucessfully run, it was able to simulate only very simplified wind histories. It proved impossible to program a more flexible simulation capacity into the model within the computational limits of IIASA computer equipment. In another application at IIASA, Somlyódy tested a multi-layer model developed by V.I. Kvon (Institute of Hydrodynamics, USSR Academy of Sciences, Novosibirsk, USSR). That model was also limited to simplified wind histories that prevented the simulation of transient behavior as actually observed on the lake.

In one of the more intensive Lake Balaton circulation studies Shanahan, Harleman and Somlyódy (1981) employed a fully transient three-dimensional Ekman-type model. The study led to apparently successful calibration and verification. Subsequent study has shown that this model is unable to correctly determine the vertical current structure in a very shallow lake such as Balaton, however. The critical factor to accurately predicting motion in very shallow lakes appears to be a non-linear bottom friction law. Linear approximations, as used by Shanahan, Harleman and Somlyódy, inadequately account for this significant force in shallow lakes (Shanahan and Harleman, 1982). All of the early modeling attempts led to predicted horizontal current patterns which were dissimilar to those observed in Lake Balaton. Thus, they are flawed not only by the practical limitations cited above, but uncertain validity as well.

One-Dimensional Lake Balaton Model

Two programs eventually led to successful circulation models for Lake Balaton: the one-dimensional model by Somlyódy (1982) and the two-dimensional model by Shanahan and Harleman (1982).

The one-dimensional model includes all computational features of higher dimensional circulation models. The formulation includes the force of wind upon the water surface and a non-linear bottom friction force which varies as a function of the average cross section velocity. The model equations are developed from the equations of motion, integrated over the lake cross section. An implicit finite difference technique is used to solve the resulting equations for flow and water surface displacement. (See Somlyódy and Virtanen, 1982 for full details on the model formulation and solution technique.) In addition to geometrical and wind data, the model requires two parameters: wind and bottom friction coefficients. These were determined by Somlyódy (1982) by calibration against the empirical relations developed by Muszkalay (1973). Once calibrated, the model was verified in simulations of selected historical events from Muszkalay's data base. Typical simulation results are shown in Figure 1. The verification simulations varied from highly successful to extremely poor, depending largely upon the nature of the wind conditions during the period simulated. This variation in results led Somlyódy to investigate the sensitivity of the model to the parameter values and input data. His very interesting findings are discussed further in the following paragraphs.

Testing against historical events showed that best results were consistently achieved when the wind was aligned with the long axis of the lake. In contrast, when the wind was directed across the lake the results were very inaccurate, with the predicted currents often opposite in direction to those observed by Muszkalay. The reasons for these errors are clear. First is the inherent error in measuring the wind direction. That error is due to both turbulent fluctuation and the finite resolution and accuracy of measurement instruments. Small errors in the wind direction result in large errors in the longitudinal component of the wind force when the wind is nearly perpendicular to the lake's long axis. The same uncertainty in wind direction will cause very little error in winds which are directed along the lake, however. Compounding this sensitivity to the wind direction, is the inherent error in measuring winds across the lake. Winds blowing across the lake, particularly those from the prevailing northerly direction, are significantly affected by the mountains to

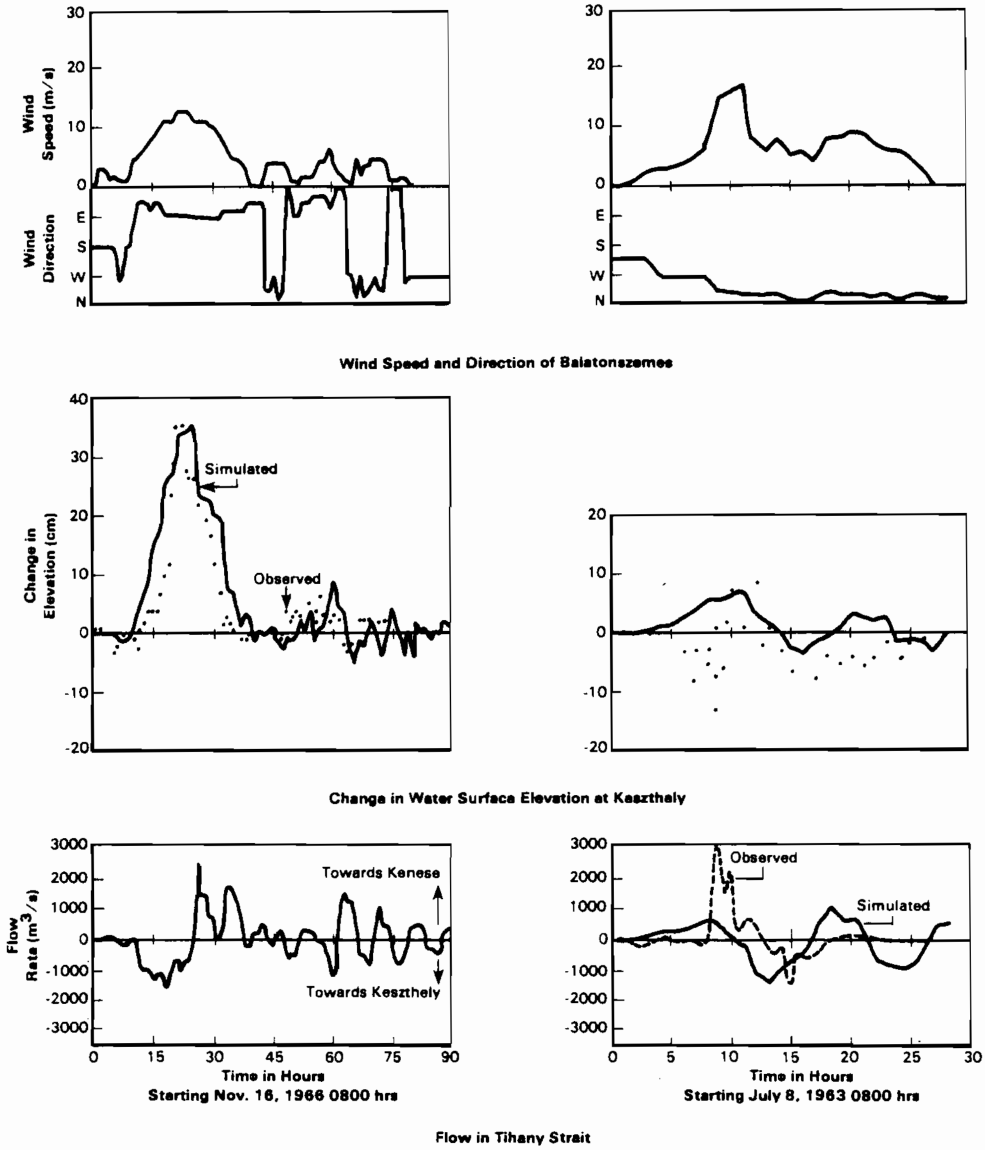

Figure 1 Typical Simulation Results with One-Dimensional Model (Somlyódy, 1982)

the north of the lake. Locally, the hills produce sheltering and deflecting effects which can produce large local differences in wind direction. With a limited number of wind recording stations, it is impossible to accurately resolve such spatial variation in the wind field.

Somlyódy (1982) performed experiments in an effort to characterize the nature of the model uncertainty and sensitivity. A series of parameter sensitivity experiments evaluated the dependence of the results upon the values of the wind stress and bottom friction model parameters. These showed that the model was only moderately sensitive to the value of the bottom friction parameter and linearly dependent upon the wind drag coefficient. The stronger dependence upon the wind drag coefficient is not surprising since the wind is the driving force for all motion in the model.

The technique of Monte Carlo simulation was employed by Somlyódy to evaluate the effect of input data uncertainty upon the model results. As the least certain of the model inputs, wind direction was selected as the parameter to be varied in the simulations. The procedure followed was to vary the direction randomly with a uniform distribution over a range of from 45° to 67.5°. Monte Carlo simulations were performed for selected events for comparison with the deterministic verification simulations. For each event, one hundred Monte Carlo trials were performed. In each trial, a wind direction error to be added to the observed direction history was selected and the simulation was performed. The results from the one hundred histories were then statistically analyzed to determine the mean, extremes and standard deviation at each point in the history. Somlyódy's results (Figure 2) illustrate that the model is considerably more sensitive to input data uncertainty than to parameter uncertainty. Further, the stochastic simulation results show better agreement with the observed behavior of the lake than do

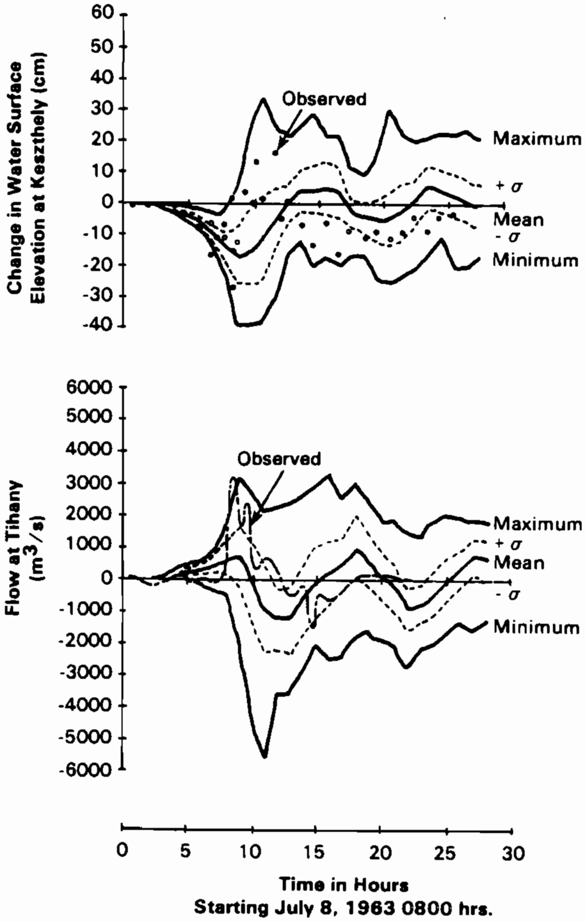

Figure 2 Effect of Wind Data Uncertainty on One-Dimensional Model Results (Somlyódy, 1982)

the deterministic simulations. The application of Monte Carlo simulation to lake circulation modeling is a new technique with promising results: it succeeded in Somlyódy's Lake Balaton studies to extract information on the fundamental behavior of the model and to accomplish a more comprehensive validation of the model.

Two-Dimensional Lake Balaton Model

The two-dimensional model, developed by Shanahan and Harleman (1982), is a fully transient model based upon the linearized equations of motion for depth-averaged horizontal flow. The model solves for displacement of the water surface and flow in the two coordinate directions as functions of time and horizontal space. The model permits a time-varying wind field to be specified and accounts for the non-linear force of friction at the lake bottom. An explicit finite difference technique is employed to solve the equations.

The parameters of the two-dimensional model are the same as those of the one-dimensional: the coefficients for bottom friction and wind surface drag. The two-dimensional model was calibrated against Muszkalay's (1973) empirical formulae, with calibration resulting in the same parameter values as found for the one-dimensional model. The model was found similarly sensitive to the parameter values and to the uncertainty of the wind input data. Monte Carlo simulations were not performed as they were for the one-dimensional model, however testing of the two-dimensional model with varied wind direction inputs illustrated the same basic sensitivity. An important result from the two-dimensional model was the prediction of horizontal current patterns similar in character to those observed in Lake Balaton (Figure 3). Previously tested two and three-dimensional models had been unable to replicate the many horizontal gyres observed.

The purpose in developing the two-dimensional circulation model was to compute mass transport information for input to a one-dimensional circulation water quality model of Lake Balaton (Shanahan and Harleman, 1982). For the purposes of the one-dimensional water quality model, two types of transport may be defined: advective transport, determined as the integral of the motion across any given cross section of the lake, and dispersive transport, an apparent longitudinal mixing due to the non-uniformity of velocity within the cross section. Shanahan and Harleman (1982) used the two-dimensional model to explore the character of these transports in Lake Balaton and to evaluate their influence upon the lake's water quality.

Dispersive mixing is conventionally represented via the dispersion coefficient in well-known analogy with the coefficients for molecular or turbulent diffusion. Shanahan and Harleman (1982) propose a procedure for the calculation of the one-dimensional dispersion coefficient from the longitudinal and lateral flow information supplied by the two-dimensional circulation model. The theoretical development is relatively lengthy and will not be given here; the interested reader is referred to Shanahan and Harlemen (1982) for details. In short, the procedure is an adaption of that presented by Holley, Harleman and Fischer (1970) for oscillatory channel flow. Shanahan and Harleman's changes are to consider the situation when lateral (secondary) currents are a more significant lateral mixing mechanism than the lateral diffusion assumed to predominate by Holley et al. The procedure further accounts for the tendency of oscillatory flow to decrease the effective dispersion. In Lake Balaton, the oscillatory motion is that due to the longitudinal seiche.

The two-dimensional circulation model results were used to predict the dispersion coefficient as a function of time and longitudinal distance. Results from the simulation of the period July and August 1977 are shown in Figure 4 as the advective flow and dispersion coefficient at selected cross sections in the lake. Both advection and dispersion are highly transient and spatially variable. Dispersion is greatest during occasional wind storms and at locations

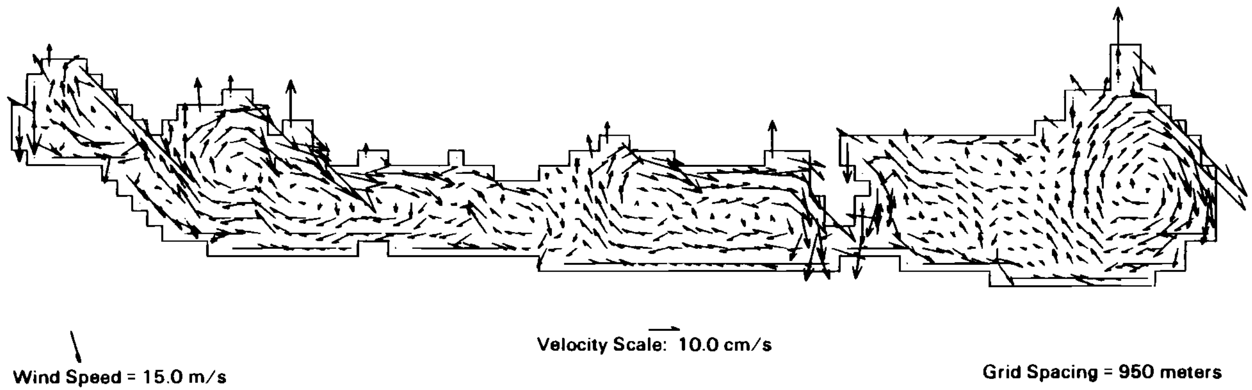

Figure 3 Horizontal Current Pattern Predicted by Two-Dimensional Model (Shanahan and Harleman, 1982)

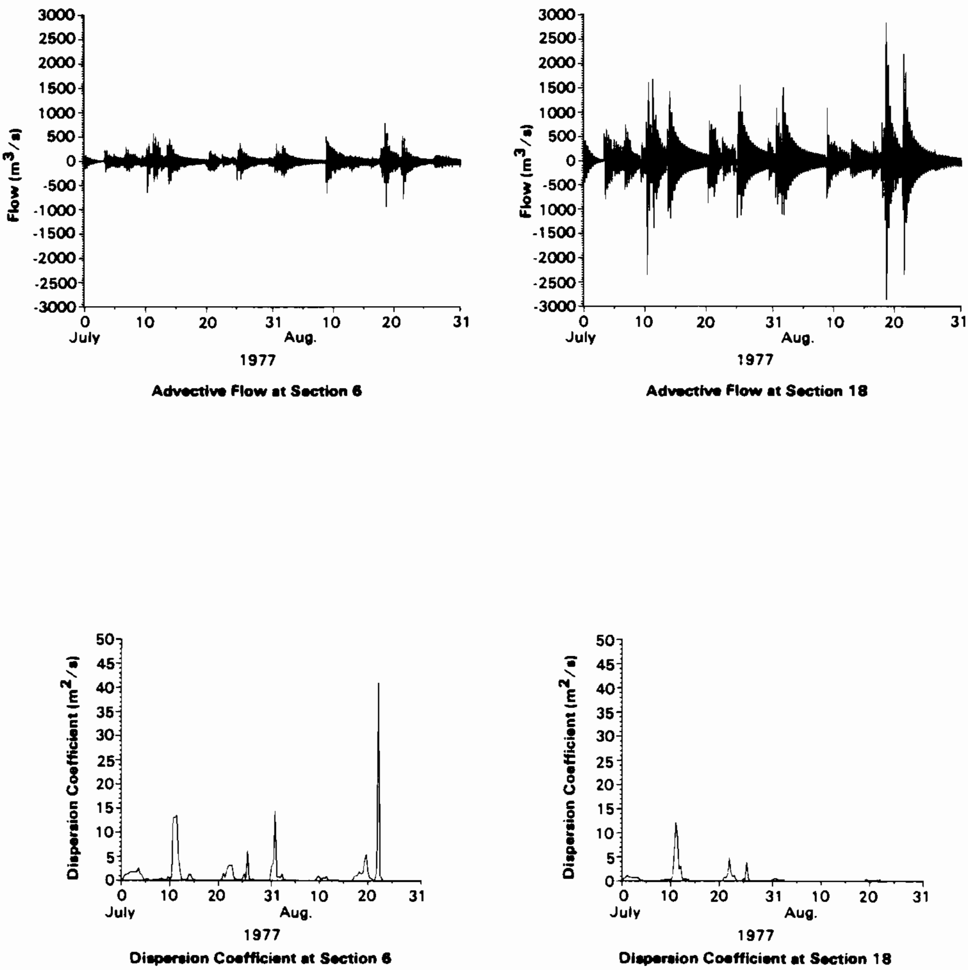

Figure 4 Advective Flow and Dispersion Coefficient Predicted by Two-Dimensional Model (Shanahan and Harleman, 1982)

where the lake's cross-sectional geometry changes abruptly. The spatial variation is indicated more clearly in Figure 5, which shows the average dispersion coefficient along the lake during the two-month simulation period.

ROLE OF MASS TRANSPORT AND CIRCULATION IN LAKE WATER QUALITY

The preceding detailed discussion of hydrodynamics and circulation modeling may strike some as out of place in a meeting dedicated to issues of eutrophication. However, circulation plays an important role in establishing the spatial variation of water quality and trophic status in a large lake such as Balaton. The Lake Balaton Case Study served as the impetus for a number of investigations into the role of circulation and mass transport in lake eutrophication and computer modeling of lake water quality.

Two major classes of phenomena influence lake water quality and trophic status. One main influence is biochemical reaction — the transformation of dissolved or suspended constituents due to biological and physiochemical reactions. The action of these processes is determined by the character of the lake chemistry and ecosystem, and their response to such external forcing functions as nutrient influx and meteorological conditions. A second major influence is mass transport, due to both advective and dispersive modes of transport. These processes affect lake water quality by moving and mixing masses of water containing dissolved and suspended constituents. Mass transport is due primarily to wind-driven circulation and hydrologic through-flow, which in turn are driven by wind force upon the lake and the distribution and magnitude of inflow and outflow to the lake.

In previous lake eutrophication modeling research, the emphasis has been upon biochemical kinetics. Typically, a simplified spatial structure and mass transport formulation has been employed in the modeling framework. In contrast, the model would simultaneously

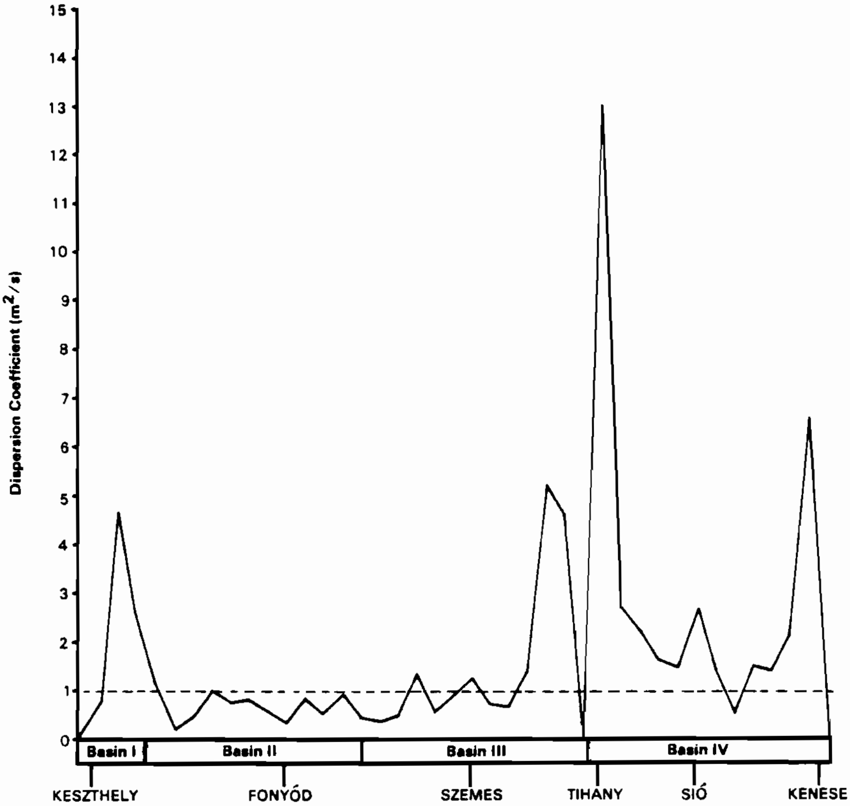

Figure 5 Spatial Distribution of Predicted Dispersion Coefficient for July and August, 1977 (Shanahan and Harleman, 1982)

include sophisticated and complicated biochemical kinetic formulations. The most common spatial structure in the lake modeling literature is the multiple-box model, in which the lake is considered as a few fully-mixed segments ("boxes") of relatively large volume.

Research performed at the Massachusetts Institute of Technology in cooperation with the Lake Balaton Case Study examined the validity of the multiple-box formulation (Shanahan and Harleman, 1982). Shanahan and Harleman's investigation included an analytical examination of this model, which is based upon a discrete formulation. As an alternative, Harleman and Shanahan considered the essentially continuous models derived from finite difference approximations of the one-dimensional advective dispersion equation. Their findings lead to a number of conclusions. Most importantly, they illustrate that the multiple-box formulation includes an implicit dispersion in addition to any explicit dispersion specified by the modeler. This dispersion increases as the number of boxes included in the model decreases. Thus, three possible situations may exist when the number of model boxes is chosen a priori. If the number of boxes is too few, the model will be over-dispersive by virtue of the model's implicit dispersion. In this situation, the internal mixing characteristics of the actual lake cannot be accurately represented by the model. In the second situation, the number of boxes fortuitously carries an implicitly dispersion which approximates the dispersion of the actual lake. Here, the model can adequately portray the dispersive transport of the lake by using the implicit dispersion as a surrogate for the actual dispersion. In the third situation, the number of boxes is sufficiently large that the implicit dispersion is less than that of the actual lake. To compensate, equal but opposite exchange flows between adjacent boxes must be specified in order to increase the internal mixing within the model. Typically, the magnitude of such exchange flows is determined by calibration. In the latter two of these three situations, an adequate model of the lake is possible, while in the first situation mass transport in the lake cannot be properly modeled. Regardless of the specific situation, there is a fundamental weakness in the multiple-box formulation due to the vagaries of the implicit dispersion. The consequence of the implicit dispersion is diminished control over the mixing characteristics of the multiple-box model. The large implicit dispersion confounds the modeler's selection of box size and boundaries and precludes direct specification of dispersion or model exchange flow based on observed hydrodynamics.

In contrast, a properly constructed finite difference model is characterized by a minimum of intrinsic dispersion. So-called numerical dispersion is possible in such models, however proper finite differencing techniques (such as second-order accurate central differences) reduce such dispersion to a negligible level. This makes possible, indeed requires, the modeler to specify a longitudinal dispersion coefficient based upon the physical characteristics of the lake and its hydrodynamics. Unlike the multiple-box situation, model mixing is a model-independent parameter which can be specified directly from observation or other knowledge of the physical system.

The conclusions from Shanahan and Harleman's analysis are that the box model is an acceptable lake water quality model only if certain conditions are met. First, the number of model boxes and their geometry must be chosen with care to ensure that the implicit dispersivity of the model is within reason. Second, there must be adequate field data -- preferably for a conservative tracer substance -- with which to calibrate the model exchange flow. Shanahan and Harleman also conclude that a finite difference model of the advection-dispersion equation is a more straightforward alternative to the box model. In the finite difference model, dispersive mass transport is essentially independent of the model construction, thus it can be specified from hydrodynamic information rather than by calibration against water quality data.

LAKE BALATON MASS TRANSPORT STUDIES

The role of mass transport in eutrophication modeling was examined in two of the investigations contributing to the Lake Balaton

Case Study. In the first, van Straten (1980) considered the influence of dispersive transport in a four-box model of the lake as part of a broader study into model and parameter uncertainty in eutrophication modeling. He developed a model calibration procedure to explicitly account for the various uncertainties which affect the model. The procedure allows for these factors by first defining an acceptable range of model results for historical periods based on the incomplete and uncertain observation data for those periods. Then, bounds are placed on the possible values for model parameters based on literature values. With these definitions, a Monte Carlo simulation is performed, selecting parameter values randomly from within the defined bounds and recording those parameter combinations which yield model results within the acceptable response range. van Straten employed this formalized calibration procedure to investigate alternative model formulations and to evaluate the sources of model uncertainty.

The calibration parameters included by van Straten were a number of biochemical parameters and rate constants, and a return velocity, the velocity associated with inter-box exchange flow. Multiplying the return velocity by the cross-sectional area of the interface between adjacent boxes gives the exchange flow between the boxes. Thus, the return velocity is a measure of the dispersion specified for the box model. van Straten found through his model calibration procedure that the acceptable range of the return velocity was between 0.2 and 2.4 mm/sec, but that the values of the other (biochemical) model parameters were sensitive to the return velocity value. Because the calibration procedure included both biochemical and mass transport parameters, it is difficult to draw specific conclusions concerning the influence of mass transport. Nevertheless, the sensitivity of the other model parameters to the value given the exchange velocity indicates the importance of mass transport to model behavior.

van Straten's investigation dealing with a four-box model prompted Shanahan and Harleman (1982) to consider a comparison with a finite difference model. They develop a one-dimensional finite difference model of the lake phosphorus cycle employing van Straten's biochemical submodel as a basis. Shanahan and Harleman's model differs from van Straten's in its more detailed spatial representation and its specification of mass transport. Dispersive mass transport is specified using the dispersion coefficients calculated from the circulation model results as described above. Other than these differences in spatial and mass transport formulation, the Shanahan and Harleman model is essentially identical to van Straten's.

Shanahan and Harleman compared the two model formulations in simulations of Lake Balaton for the year 1977. The simulations were made for the 250-day period beginning February 28, 1977. A complete discussion of the input data, simulation conditions and other details is given by Shanahan and Harleman (1982). Important assumptions were that the back-and-forth motion of the seiche produced such a short excursion that it did not contribute a significant component to advective mass transport, and that circulation during July and August of 1977 was sufficiently representative of the entire year to be used repeatedly as the input to the water quality model simulations. A more complete discussion of these assumptions and their rationales is given by Shanahan and Harleman.

Model results are contrasted in Figure 6. The figure shows the spatial distribution of predicted total and algal phosphorus concentration on August 4, a time when phytoplankton was near its peak summer concentration. The figure includes the results from the one-dimensional model and from the four-box model with and without exchange flow. The effect of the box model's implicit dispersion is clear: local concentration gradients and peaks are flattened in the box model results relative to the finite difference model. The effect is particularly noticeable at Keszthely Bay. Careful comparison of the results shows, however, that the four-box model without exchange flow is generally able to capture the lake-wide longitudinal gradients observed in the one-dimensional results.

The implicit dispersion in the four-box model varies along the lake as the box size varies. Following a definition of the local Peclet Number given by Zvirin and Shinnar (1976), the equivalent

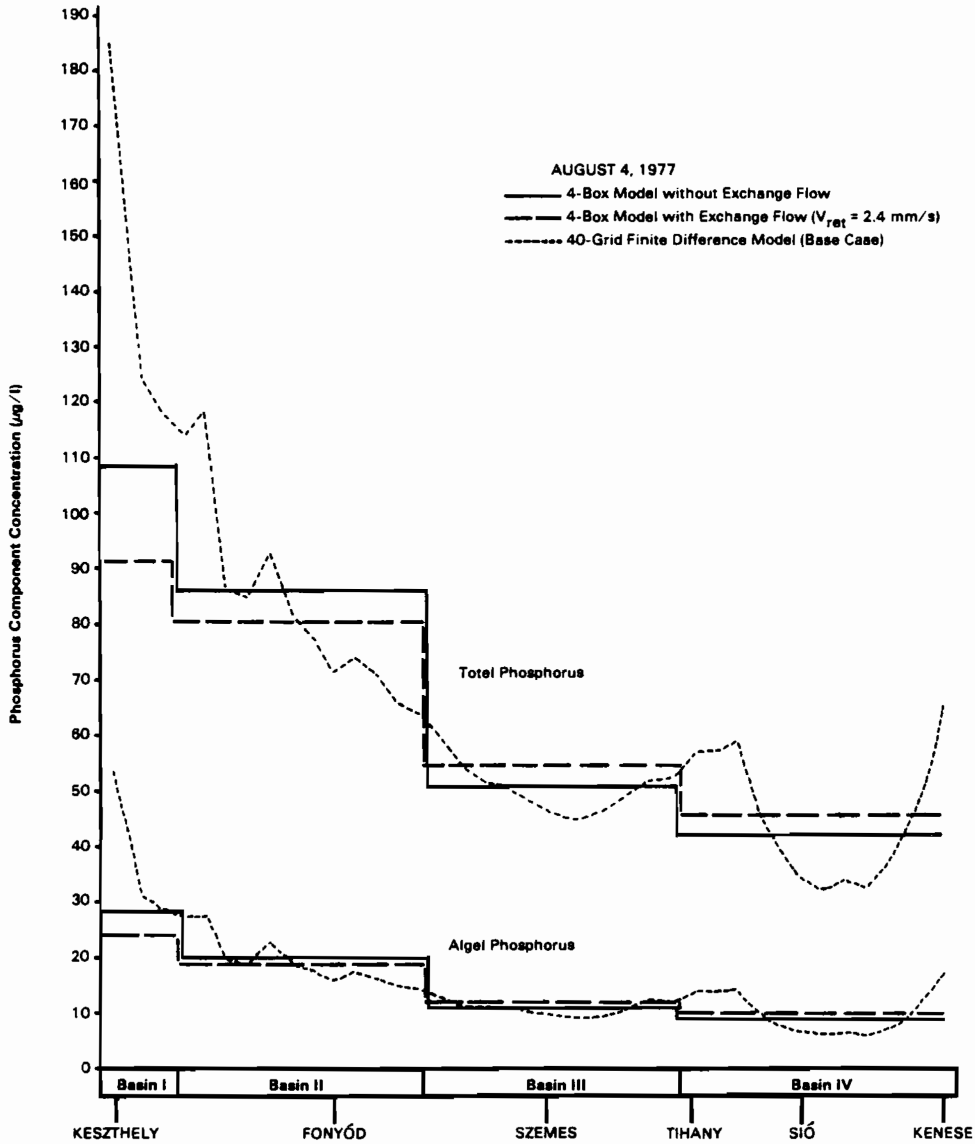

Figure 6   Comparison of Four-Box and One-Dimensional Model Results
(Shanahan and Harleman, 1982)

dispersion of the box model without exchange flow is calculated to be 1.4, 4.5, 4.7 and 4.8 m^2/s respectively in the boxes from Keszthely Bay eastward. These dispersivities agree reasonably with those determined from the circulation model results (Figure 5). If exchange flow is included, the equivalent di•persion is much higher (5.9 to 37.7 m^2/s), thus explaining the greater discrepancy between the four-box model with exchange flow and the one-dimensional model.

In summary, the contrast between the one-dimensional and four-box model results shows that the four-box model without exchange flow is an adequate model of Lake Balaton's mass transport characteristics. According to our earlier discussion of the three possible modeling regimes for multiple-box models, the Lake Balaton four-box model is an example of the second situation wherein the implicit dispersion of the model is a fair approximation to that of the actual system. Despite this ability to capture lake-wide gradients, the model comparison also shows that the box model neglects many local peaks and other spatial details predicted in the one-dimensional concentration distribution. The character of the one-dimensional model predictions is corroborated by comparison with field observations which verify that local concentration peaks occur along the lake (Figure 7). However, the field data fail to verify the concentration magnitudes predicted by either the four-box or one-dimensional models. Correction of this shortcoming must await additional basic study of phosphorus dynamics in the lake and subsequent proper representation of those dynamics in biochemical models.

AREAS FOR FUTURE RESEARCH

Data Uncertainty and Model Sensitivity of Circulation Models

The results of Somlyódy's (1982) studies with the one-dimensional circulation model emphasize the model's severe sensitivity to the wind data input. Otherwise, his studies and the two-dimensional modeling

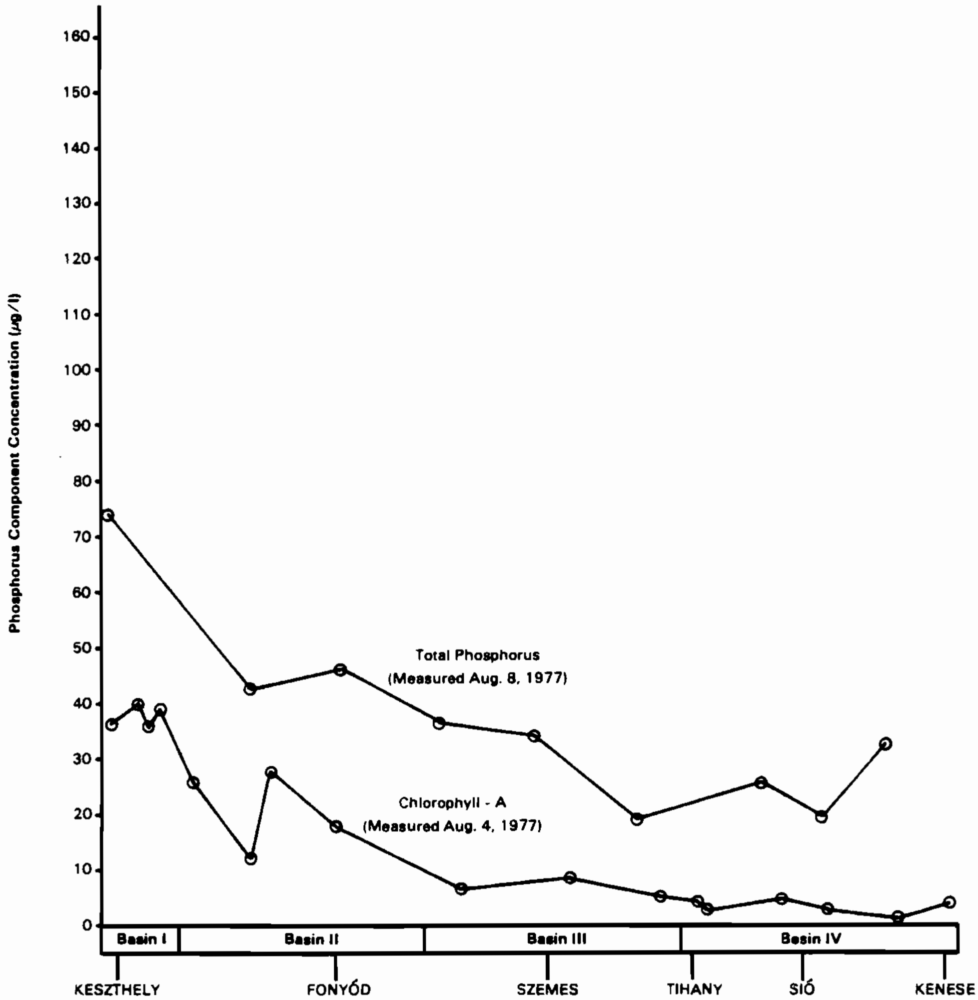

Figure 7 Field Observations of Total Phosphorus and Algal Phosphorus in Lake Balaton - August, 1977. Values of Algal Phosphorus are Based Upon the Assumption that the Algal Phosphorus Concentration is One-Half the Observed Chlorophyll-a Concentration.

studies by Shanahan and Harleman (1982) show that the parameter calibration and predictions of the model are basically sound. Thus, there is a clear conclusion that model sensitivity is driven by input data uncertainty rather than parameter or model uncertainty.

This conclusion calls for future research into the question of wind data uncertainty. The technique developed by Platzman (1963) is in wide use to account for the spatial variation in wind speed and direction. However, there is no theory or procedure to compensate for a sparse observation network. Given that wind data sparcity is inevitable, there is a need to systematically and rigorously identify its effect on circulation modeling. Somlyódy's work provides a first step in identifying data uncertainty, however more rigorous and quantified determinations should be pursued. The effect of data sparcity and the augmentation of sparse observations are topics in stochastic hydrology and control theory: cross-fertilization with these fields should prove valuable.

Spatial Current Structure

As discussed above, steady-state theory predicts vertical current structure under wind-driven conditions in a closed water body. The research on Lake Balaton leaves open the question of this widely-supposed model's validity and applicability to real lakes under transient conditions. For example, the time to reach such a steady-state depends upon the fetch length. In a large lake, the condition may be rarely seen since winds change too quickly and seiche motion will occur instead. As well, lateral current structures will be created in a shallow lake between sectors of different depth: lateral circulation may then dominate vertical circulation. These questions are all important to water quality and eutrophication modeling since the structure of the current field determines the mechanism and strength of hydrodynamic dispersion in the lake.

Unfortunately, the data available from the Lake Balaton studies leaves the dominant circulation unclearly identified. The three-dimensional modeling studies by Shanahan, Harleman and Somlyódy

(1981) were thwarted by the difficulties in defining the vertical eddy viscosity function and the bottom boundary condition in very shallow lakes. An area for future research would be the development of three-dimensional models capable of capturing the difficult conditions of the very shallow lake. The expense of modeling in three-dimensions is sufficiently high that the model resulting from such study would be a research tool only: it would be too expensive for practical simulation of seasonal or longer periods. Development of a three-dimensional shallow lake model would necessarily require detailed field data for verification and calibration; extensive field studies would be a required part of the research and would also serve to answer questions about current structure in shallow lakes.

Dispersive Transport

Shanahan and Harleman (1982) stress the importance of proper attention to dispersive transport in the construction of lake water quality models. As an alternative to the common multiple-box model approach, they employ circulation model results to compute the longitudinal dispersion coefficient which in turn is input to a one-dimensional finite difference water quality model. A key link in this procedure is the method they propose for computation of the dispersion coefficient. However, the method is based only on theoretical considerations and was not proven through field or laboratory studies. Before the technique can be applied in confidence it should be tested and verified.

This leads to our last recommendation for future research: additional studies to confirm or reject the dispersion calculation method. Field or laboratory studies would likely yield the most robust test of the model. However, a detailed stream-tube dispersion model could possibly be used to test the model more simply and inexpensively. The value of these studies, regardless of the method employed, could be to confirm a technique which allows mass transport parameters to be isolated from the already complex task of eutrophication model verification. If valid, the method allows dispersion to be determined from hydrodynamic information independently of the water quality modeling process.

SUMMARY

The Lake Balaton Case Study has sponsored or collaborated in a number of research studies concerned with the role of wind-driven circulation and mass transport in lake eutrophication and water quality. The investigations have concluded with the successful development of one-dimensional and two-dimensional circulation models, as well as the completion of studies which link circulation model results to models of lake eutrophication. Significant contributions from the Case Study include Somlyódy's techniques and findings concerning uncertainty in circulation modeling, and Shanahan and Harleman's inquiries into fundamental questions of lake model structure and proper representation of mass transport. The unsuccessful portions of the study contribute as well. Particularly, the early attempts in developing a circulation model indicate the difficulties in modeling very shallow lakes, the importance of the bottom friction boundary conditions to such models, and the detail and expense inherent in developing models of vertical current structure in very shallow lakes. These and other findings from the Case Study have pointed to a number of areas for future research. Considerably more inquiry into the character of wind data uncertainty is necessary to develop techniques which compensate for data inadequacy when modeling circulation. Studies to confirm the dispersion calculation procedure proposed by Shanahan and Harleman are also suggested for future research, as is more fundamental research into the spatial character of wind-driven currents in very shallow lakes.

ACKNOWLEDGEMENTS

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LAKE EUTROPHICATION MODELS

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INTRODUCTION

Lake eutrophication is the process of excess fertilization with plant nutrients, most notably phosphorus and nitrogen components, as a consequence of the development of human activities in the lake's watershed. The subject of this report is the relation between the nutrient input and the response, in particular the biological response, of the lake. Since the physical, chemical and biological processes in a lake are extremely complex, modelling has been used frequently in the recent decade as a tool to organize the available information in a systematic and useful way. Hence, a more precise description of this report's subject is: lake eutrophication modelling. The objective is to explore what we have gained from modelling exercises thus far, what are the weak spots and what we can conclude about the expected effects of lake eutrophication abatement programs. Though much of what will be said holds for lakes in general, most of the discussion will focus upon Lake Balaton as the particular example of interest during the workshop for which this paper was prepared.

In practice there may be various reasons why one would resort to modelling when studying a lake's eutrophication process. Among the most important objectives for modelling are:

(i) to improve our comprehension of the complexity of the real world in an integrated way, thus allowing us to see the overall effects of the various subprocesses on the system as a whole.

(ii) to detect, by comparison of model results and field data, black spots in our knowledge, which may lead subsequently to guide-lines for measurement programs and further research.

(iii) to provide a tool for judging the effectiveness of management options under consideration for eutrophication abatement.

When reviewing the activities related to the IIASA Lake Balaton Case Study we may state that the study has served all three purposes, with a gradual shift towards management application (item iii) now that the study reaches its completion. Since management is the companion theme of this closing workshop most of what follows is centered around an evalution of the type of answers that can be expected from eutrophication models for management purposes.

Loosely speaking, in lake eutrophication modelling two broad classes may be distinguished:

(i) statistical models, i.e. models based upon correlations between a limited number of aggregated variables, collected for a large number of lakes.

(ii) dynamical models, i.e. models based upon mass balance considerations, mostly in the form of differential equations with time as the independant variable.

A third class, which has not been explored extensively, is equally based upon the mass balance concept, but only considers a steady state or series of steady states. Although steady state is never really reached in a natural situation the technique may be useful in some cases, and will be developed in this paper as a tool to analyze the major features of a dynamic model structure typical for a large number of eutrophication models.

STATISTICAL MODELLING

In the early seventies, lake eutrophication was recognized as a problem which occurred world-wide. This led the Organisation for Economic Cooperation and Development (OECD) to start a study to investigate this phenomenon on a global scale. All over the world scientists volunteered to collect and supply data about the lake or lakes of their interest. Such data were generally aggregated values, e.g. annual average total phosphorus concentration, annual average chlorophyll, summer average chlorophyll, maximum summer chlorophyll, etc., together with nutrient loading information and lake geomorphological and hydrological data such as average residence time and surface hydraulic load. In addition, the investigators were asked to qualify their lake(s) as oligotrophic or eutrophic. First, the information was used to elaborate a method for qualifying a lake as eutrophic or oligotrophic when hydraulic load and areal P

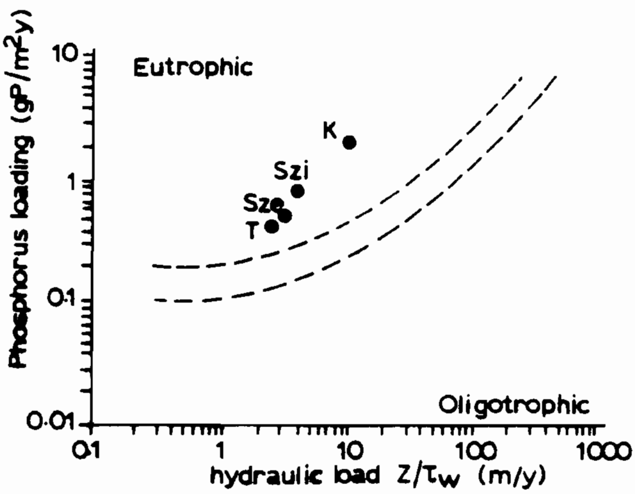


FIGURE 1: Position of the four Balaton basins in the investigator judged lake classification plot for the OECD lakes. (K=Keszthely, Szi=Szigliget, Sze=Szemes, T=Tihany/Siófok)

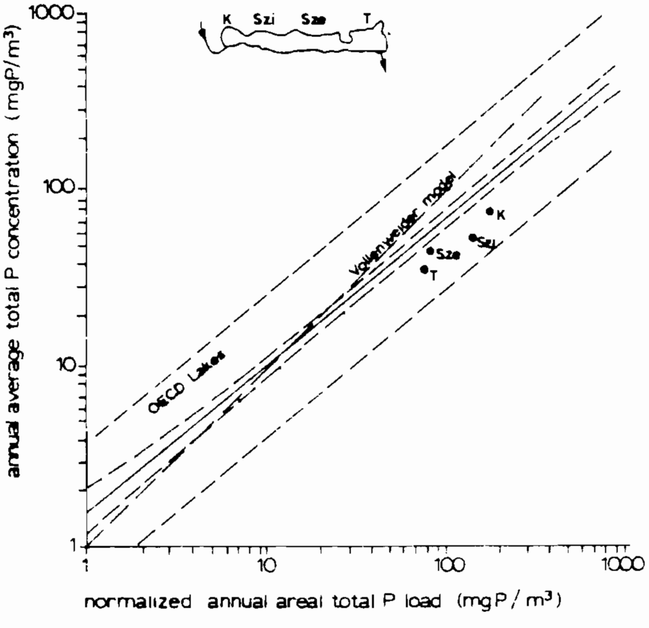


FIGURE 2: Lake total P concentration response to normalized load (defined as $Lq_s^{-1}(1+\sqrt{\tau_w})^{-1}$, OECD best fit line and 1σ and 2σ boundaries), and position of the Balaton basins. For comparison the Vollenweider 'model' line (equation 1) has also been shown.

loading were given. The result is shown in Figure 1 (adapted from Jones and Lee, 1982). In this figure also the four basins of Balaton have been plotted. We see that they can all be characterized as eutrophic, even the relatively lowly polluted Tihany/Siófok basin. Since the boundary line between eutrophic and oligotophic is based upon subjective judgement, and consequently is rather uncertain the plot has no other value than indicative, and is hardly of interest for management purposes.

More useful for management is the attempt made to related some of the observed lake variables to phosphorus loading. After numerous trials the promotor of the OECD-study, Richard A. Vollenweider, proposed the following empirical model for the relation between annual average total P concentration and loading, hydraulic load and 'filling time':

$$P_\lambda = \frac{L}{q_s} \frac{1}{1 + \sqrt{\tau_w}} \tag{1}$$

where P_λ : annual average total P concentration [mg P/m^3]

L : areal annual total P loading [mg P/m^2 y]

q_s : hydraulic load = inflow per unit surface area [m/y]

τ_w : filling time (residence time) = volume/inflow [y]

(Vollenweider and Kerekes, 1980, 1982).

Figure 2 shows the line of best fit of the OECD lakes when plotting the annual average total P concentration (the left hand side of equation (1)) versus the right hand side (sometimes refered to as the 'normalized annual total P loading', Jones and Lee, 1982). Also shown are the ranges observerd around this line of best fit. The 45 degrees line is the line that would have occurred had equation (1) been exact. We may see that this 'model' line is well within the boundaries of the empirical best fit, but since the plot is double logaritmic the deviations in concentration terms are considerable.

In Appendix I data for Lake Balaton are provided together with some simple calculations that would allow the positioning of the four Balaton basins in the plot of Figure 2. These points are shown as dots in Figure 2. It can be seen that the four basins of Balaton seem to fall well within the boundaries obtained within the OECD study. This suggests that Lake Balaton is not, in this respect, an exception, as is sometimes believed given the exceptionally large calcium content and its associated phosphorus binding capacity (cf. Park, 1978). The result of Figure 2 is the more remarkable, since the OECD line was obtained for an ensemble of lakes of which the majority was deep. In fact, when only the shallow lakes and reservoirs were incorporated a slightly better correlation could be obtained using the model

$$P_\lambda = \frac{L}{q_s} \frac{1}{1 + 2\sqrt{\tau_w}} \tag{2}$$

(Clasen, 1980). However, for lakes where τ_w is relatively short, such as Lake Balaton, no large differences are expected, and this is confirmed by Figure 3 where the four basins are figuring in a plot drawn on the basis of equation (2). So, the use of the more generalized empirical fit derived for all lakes irrespective of depth (equation 1) would be appropriate in most cases even for shallow lakes.

Because Lake Balaton falls well within the framework of the OECD studies, an important conclusion for management is apparent: on the basis of the OECD statistical result a reduction in external loading must be expected to result in a reduced annual average total P concentration in the lake. This is a significant outcome. However, since Figure 2 is plotted on a log-log scale the confidence ranges are quite wide: they correspond to a variation of a factor 2.5 above and below the best fit line. Perhaps a more direct impression of the extend of these uncertainties is obtained by plotting the annual areal load to the annual average total P for a particular lake on a linear scale. This is done in Figure 4 for the Keszthely Bay of Lake Balaton. The conclusion is quite obvious: all we can say is that the trend is clear, but more quantitative statements can hardly be made.

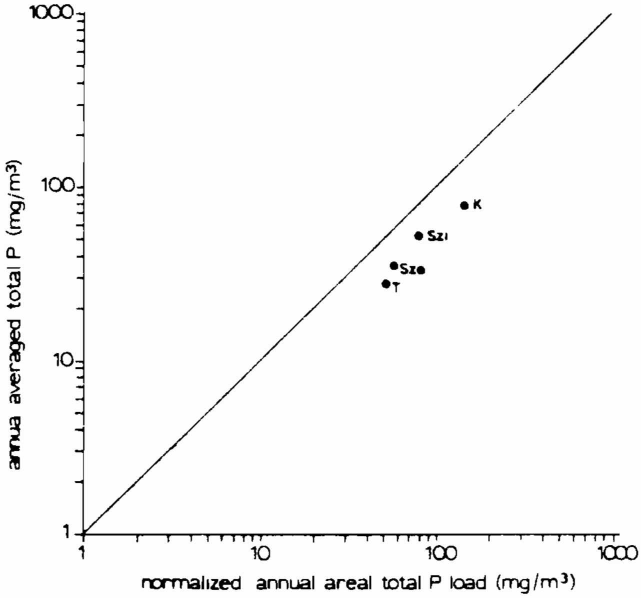


FIGURE 3: Position of the Balaton basins in the lake response plot according to the OECD Shallow Lakes and Reservoirs formula (equation 2).

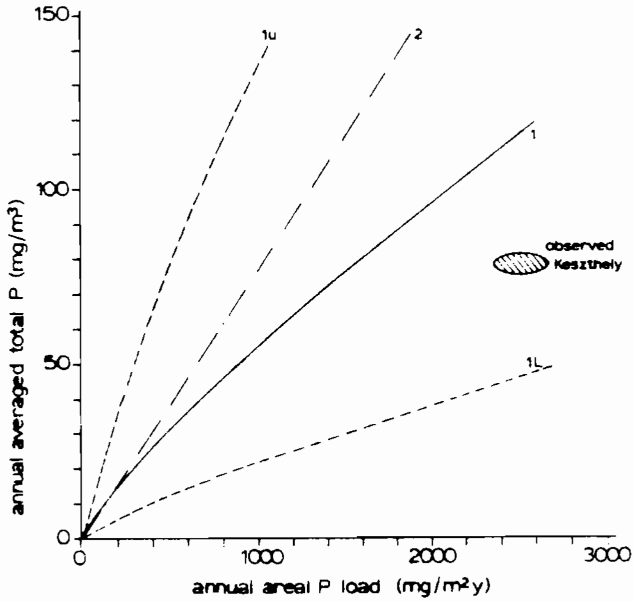


FIGURE 4: Direct relation between annual average total P and areal annual load predicted by OECD statistics for Keszthely Bay, Lake Balaton.
1: OECD best fit line; 1u,L: upper and lower bound;
2: Vollenweider model line

In lake eutrophication management total phosphorus is only an indicator for water quality. Rather, one would be interested in quantities like biomass or chlorophyll-a as an indicator for possibly objectionable algal blooms. This aspect has been dealt with within the OECD study, and in general a strong correlation was found between maximum chlorophyll-a or summer averaged chlorophyll-a and total phosphorus concentrations. Consequently, similar plots can be made to express the dependancy of these quantities with total P loading. An example is given in Figure 5. Here, again the points for the four Balaton basins have been indicated, and again similar conclusions can be drawn: reducing the external loading does reduce the observed maximum algae concentration in a lake, but, again, the uncertainty is quite large.

Finally, Figure 6 shows the trends suggested by the OECD results for the four Balaton basins on a linear scale. Note that the load is expressed here per unit basin volume. The differences in response slopes result from the different hydraulic loading and residence times for the four basins.

To conclude this section one may say that the statistical approach is a useful tool for a rough and preliminary analysis of the possible effect of management options. In cases where the management options are straight-forward, and no fine-tuning is required, at least not in the initial stages of implementation, the approach may even be entirely sufficient. It really depends upon the degree of continuity in the management alternatives, together with considerable cost differences among the alternatives, whether a more precise statement is necessary. In such instances more accurate answeres could be obtained (potentially) only by resorting to dynamic modelling.

DYNAMIC MODELLING

This approach is based upon dynamic mass balance equations, involving (approximate) mathematical descriptions for each of the processes

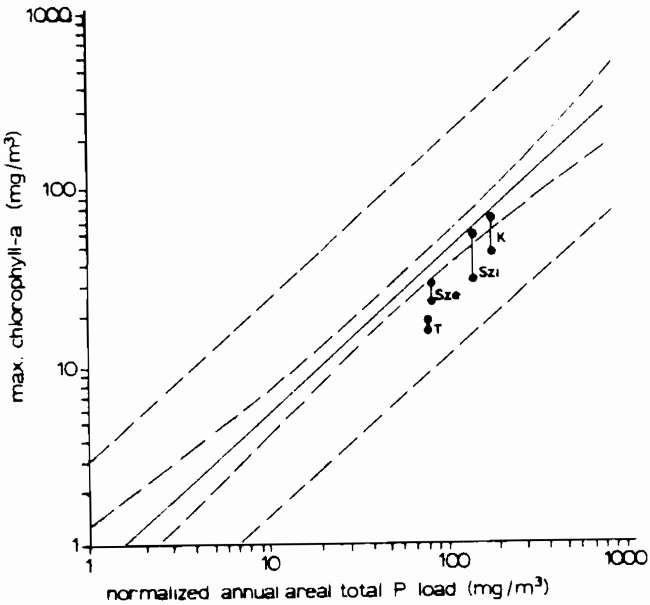


FIGURE 5: Lake maximum chlorophyll-a response to normalized phosphorus load for OECD lakes, and position of the Balaton basins (best fit and 1σ, 2σ boundaries).

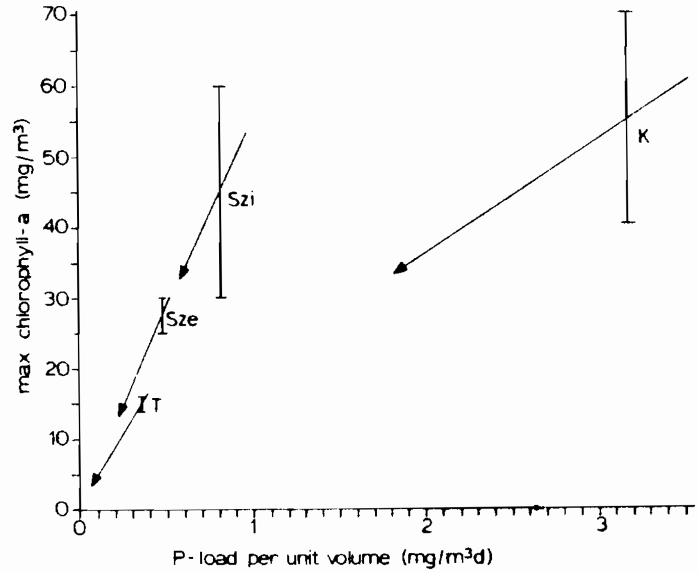


FIGURE 6: Trend in reaction of maximum chlorophyll-a to phosphorus load reduction according to OECD empirical relations.

of interest. These models have as advantages over the statistical approach that they allow for more time and space detail, that they generally consider a number of relevant variables rather than just one or two, and that they are potentially more accurate once properly calibrated. The increased spatial and temporal resolution may be a desirable property for management purposes, especially when it comes to the description of peak events which are causing most of the nuisance. The disadvantage of the dynamic modelling approach stem partly from the same items constituting its advantage: the larger detail must be bought for a large, and sometimes exhaustive, data requirement. Moreover, frequently accuracy is affected in a negative sense because the mechanisms of some of the processes incorporated in the models may not be known to a satisfactory degree. Despite these shortcomings dynamic models proved very successful in organizing and disseminating results of scientific research, and have been helpful as guidelines for further in situ investigations. And, as will be shown below, specific conclusions of management interest can be drawn, even with the limitations outlined above.

Typically, dynamic models contain three types of terms:
- hydrological and hydrodynamical transport terms.
- terms related to input, output or exchange of material through the boundaries; we will call these terms interface terms.
- terms related to chemical and biological transformations.

It should be noted that the rate of change of these processes depends upon extraneous or forcing variables, such as temperature, solar radiation, wind speed and direction etc., which means that fairly detailed records of these variables must be available if a dynamic modelling attempt is going to be successful.

The first two types of terms above will be treated very briefly here, because they are the subject of other general reports (P. Shanahan and D.R.F. Harleman, this proceedings, and L. Lijklema, P. Gelencsér and F. Szilágyi, this proceedings). The third term will be covered in somewhat more detail.

TRANSPORT TERMS

Here both transport from one end of a lake to the other end due to hydrological throughflow, as well as internal transport and mixing due to wind action or heat generated buoyancy are of interest. For shallow lakes the latter can usually be ignored. Generally, in literature, attempts are made to circumvent the difficult internal transport and mixing problem, for instance by considering the system as ideally mixed. In fact, only a few examples of full transport oriented eutrophication models can be found (e.g. Lam and Jaquet, 1976). For Balaton, a one-dimensional transport model was developed by Shanahan and Harleman (1982) where the longitudinal dispersion coefficient was derived from a 2-D hydrodynamic circulation model. Eventually, such models may be used for the purpose of simplifying the hydrodynamical structure of a system on a sound basis, in contrast to the usual practice of choosing the desired number of completely mixed control volumes by subjective judgement. In the case of Lake Balaton it turned out a posteriori that the initial and subjective choice of four boxes to represent the various basins was quite appropriate from a hydrodynamic point of view. It should be emphasized, however, that this simplification does entail a loss of spatial detail: especially in the Keszthely region the predicted biomass concentration at the outer edge of the basin is considerably higher than the fully mixed basin average. This fact may be of interest for the management, particularly in judging the results of abatement programs on the basis of field measurements which may not be fully representative for the basin average.

INTERFACE TERMS

One class of terms in this category is related to the external loading into the system. Considerable efforts are generally needed to evaluate these terms (see general report by G. Jolánkai and L. David; this proceedings) but in this respect at least no essential differences exist between dynamic and statistical modelling.

Another class of terms is associated to exchange processes, e.g. of oxygen and carbondioxide at the water surface, and the exchange of phosphorous compounds with the sediment. The latter process is extremely complex, as was pointed out by Lijklema (this proceedings). Physico-chemical processes such as adsorption or chemosorption on calcium and iron compounds are affected by oxygen conditions and pH, which in turn depend upon both the biological processes in the water columns, as well as transport and transformation processes within the sediment. In most lake eutrophication models the sediment, if at all present, is extremely simplified. There is little doubt, that the ignorance or over-simplification of the sediment is responsible to a large extent for the less satisfactorily performance of dynamic models, especially for shallow lakes, and as long as fundamental research in this field is lacking there is not much hope for improvement. On the other hand, with proper analysis based upon the limited information available now, still some useful and relevant statements can be made regarding the role of the internal sediment pool in lake restoration, as will be shown later.

TRANSFORMATION TERMS

Chemical transformations

Chemical reactions may have a significant effect upon the biological cycle and the eutrophication process of a water body. An important example is the regulation of the pH of natural waters, mainly by the (bi)carbonate equilibria and dissolution or precipitation of metal-carbonates such as $CaCO_3$. The biological cycle itself exerts an influence upon these reactions because of fixation or liberation of CO_2 (by photosynthesis and mineralisation, resp.), and so is responsible for the difference in pH between e.g. a productive water-body (pH-range 8 - 10) and a consumptive sediment (pH $<$ 7). In addition to pH-influences on the life-cycle itself, indirect effects exist such as phosphate nutrient coprecipitation with biogenic lime, which may be a significant term in the total nutrient loss to the sediment, especially in summer. Another phosphate removal mechanism may be associated with iron and calcium compound precipitates formed when river water gets into contact with lake water with a higher pH. In this light it is somewhat strange to observe that most eutrophication models published in literature ignore this field, or treat the matter in an extremely simplified way. Examples of recent developments which do include chemistry explicitely are DiToro (1976) and De Rooij (1980). No such chemical considerations have been included up to now in the eutrophication models developed for Lake Balaton, despite indications of significant chemical effects which may be deduced from the extensive data set. Further elaboration in this area, therefore, is expected to be fruitful.

Biological transformations

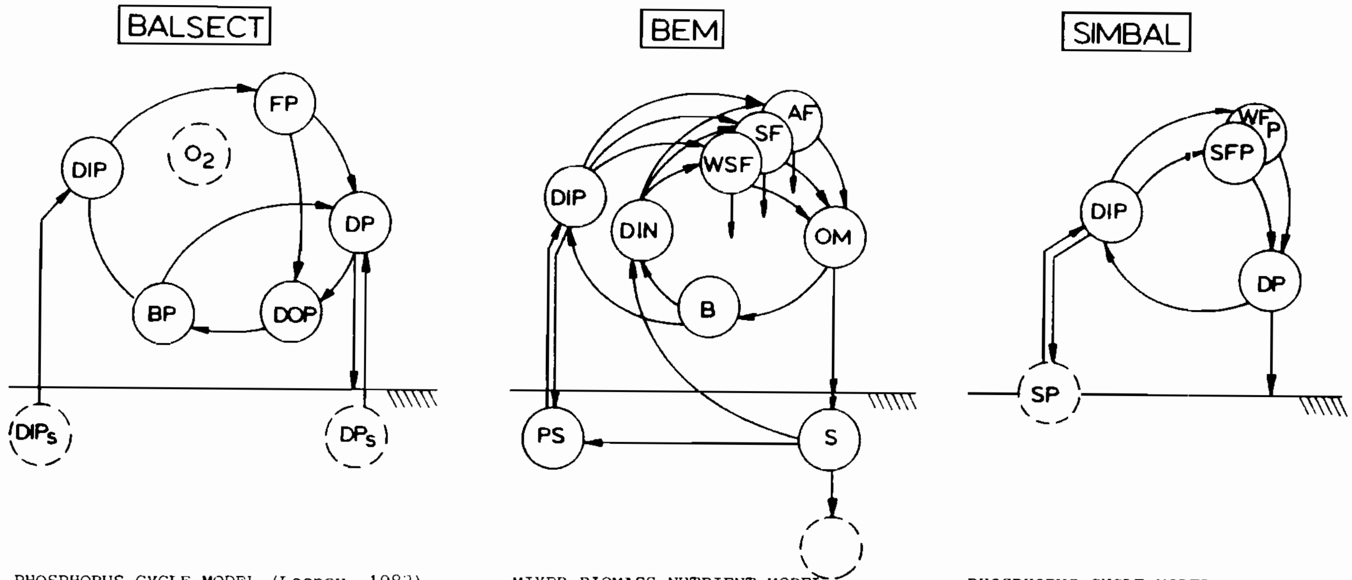
Most eutrophication models concentrate on the description of phytoplankton dynamics in the lake water throughout the year. Such a model always has associated to it a description of the cycle of nutrients, because excess supply of nutrients (most notably phosphates) is the basic cause of eutrophication. Typically, carbon assimilation by photosynthesis is accompanied by nutrient uptake in some form or another. Then, the nutrients are recycled by the sequence of processes mortality, hydrolisis and mineralization, so that most of it becomes available for uptake in the next cycle. During each loop a certain amount of material is transferred to the sediment where it is stored for shorter or longer time.

Although the principles outlined above have been accepted generally, a large number of model variants exists, especially with respect to the number of dependant variables considered ('state variables', sometimes called 'components') (e.g. Scavia and Robertson (1979), Canale (1976)), and with respect to the mathematical formulation of the sub-processes (for a review see Swartzman and Bentley, 1979). A difference in number of state variables among models arises from more or less subjective judgement about the degree of detail desired, or required to yield sufficient realism. In practice, the choice is dictated frequently by the availability of data. Examples of decisions influencing the number of state variables are: inclusion of only a single nutrient (mostly phosphorus) or of several nutrients (e.g. nitrogen, silica); consideration of grazing by zooplankton and of higher trophic levels, or not; detailed chemical component description (e.g. ortho-P, condensed P, particulate organic P, dissolved organic P, etc.) versus simplifications (e.g. only dissolved available P and phytoplankton-P); inclusion of a single or a few algae species versus multi-species modelling. Examples of differences in mathematical formulation are: mortality as a first or higher order process; multiple nutrient limitation by a multiplicative versus a minimum-rule formula; growth-rate light dependancy by a Steele or Smith relation, etc.

An important fundamental distinction between models is related to the problem how measures of biomass (biomass weight, algal counts, chlorophyll-a) are related to nutrient content. The large majority of models has used the so-called 'constant cell stoechiometry', that is, the ratio of biomass to nutrient content is a fixed constant, not varying in time. In these type of models no distinction is made between algal growth and nutrient uptake: they are strickly coupled. This allows to express the phytoplankton concentration in terms of nutrients directly, and so leads to a reduction in state variables needed. However, this advantage of simplicity is bought at the expense of a less realistic description, because the nutrient content of the algae is known to vary quite considerably. Apart from the dynamic implications this fact hampers the interpretation of constant cell stoechiometry models, since comparison with field data, such as chlorophyll-a, requires a conversion from calculated phytoplankton nutrients with a conversion factor which is not known, and, moreover, variable in time. Representations which overcome this difficulty, and which are probably also more correct dynamically are known as 'variable cell stoechiometry' or 'internal cell quota' models (Bierman et al., 1973, Bierman, 1976). In these models algal growth and nutrient uptake rate are not coupled directly.

Consequently, more state variables and system parameters have to be introduced because biomass and internal nutrients must be monitored simultaneously. Thus, these models tend to be much more complex than constant stoechiometry models. It should be noted, however, that this disadvantage can be mitigated somewhat, as was demonstrated in an analysis by Di Toro (1980), because the relatively fast response of the internal cell nutrient concentration to external concentration changes allows for a fair computational simplification.

For Lake Balaton, three phytoplankton-dynamics models have been developed in the frame of the IIASA case study, each of them of the constant stoechiometry type. The models are BALSECT ('Balaton Sector Model', Leonov, 1980, 1982), BEM ('Balaton Eutrophication Modellers Group Model', Kutas and Herodek, 1982) and SIMBAL ('Simple Balaton Model', Van Straten, 1980). A schematic overview of the structures is shown in Figure 7.
BALSECT is a phosphorus cycle model. Phosphorus is taken up from the dissolved inorganic P pool to form phytoplankton P. Phytoplankton P then moves into detritus P by mortality and into dissolved organic P by direct excretion. Dissolved organic P is mineralized by uptake by bacteria and subsequent excretion, whereas part of the bacteria P is reentering the detritus P pool by mortality. Of the three models this model has the most detailed description of the various organic P components. In contrast, SIMBAL assumes that these three components can be simplified into just one, and thus assumes that the internal transformations are either relatively fast or insignificant. Of course, some problem arises in this case because settling in SIMBAL works upon the total detritus pool, and although a correction on the settling rate is made to account for the proportion dissolved material, the dynamics of this presentation is not exactly the same as in BALSECT. A similar problem arises in the BEM model, where an 'organic material' pool is recognized, containing both particulate and dissolved material. The BEM model is not a P-cycle model, but a mixed biomass-P-N model. However, because of the constant stoechiometry a conversion would have been possible to yield



PHOSPHORUS CYCLE MODEL (Leonov, 1982)

FP : Phytoplankton P
DP : Particulate detritus P
DOP: Dissolved organic P
BP : Bacteria P
DIP: Dissolved inorganic P

DP_S: Detritus P in the sediment
DIP_S: Dissolved inorganic P in sediment

MIXED BIOMASS-NUTRIENT MODEL (Kutas and Herodek, 1982)

AF : Autumn phytoplankton biomass
SF : Summer phytoplankton biomass
WSF: Winter and spring phytoplankton
OM : Organic material (detritus)
B : Bacteria biomass
DIN: Dissolved inorganic N
DIP: Dissolved inorganic P

PS : Exchangeable P in sediment
S : Organic material in sediment

PHOSPHORUS CYCLE MODEL (Van Straten, 1980)

WFP: Cold water phytoplankton P
SFP: Warm water phytoplankton P
DP : Detritus P
DIP: Dissolved inorganic P

SP : Adsorbed exchangeable P

FIGURE 7: Comparison of structure of Balaton phytoplankton-dynamics models.

a P-cycle model. The most remarkable difference with respect to BALSECT and SIMBAL is the inclusion of a nitrogen nutrient as a state variable. Thus, possible nitrogen limitation can be covered with BEM.

Another difference between the models lies in the degree of detail with respect to phytoplankton. BEM is the most complete, covering both summer and autumn phytoplankton (i.e. mainly blue-greens), as well as winter/spring phytoplankton (mostly diatoms). SIMBAL considers both warm water species dominated communities ('summer phytoplankton, most notably Ceratium hirundinella) and cold water species dominated communities ('winter phytoplankton'). The distinction between these two groups was made upon the basis of field data on primary production, from which a fairly adequate estimate of the necessary additional parameters could be made (Van Straten and Herodek, 1982). Thus, no unsupported detail was incorporated in the model. BALSECT has the most simple algal compartment. Succession over the season is largely covered by introducing a two-peak temperature function for algal growth, a construction primarily employed in SIMBAL but abandoned later. It should be noted that a comparison of BALSECT results with field data was made by comparing the sum of the calculated algal P and non-living particulate P with measured particulate organic P. This procedure, although correct as such, may mask the seasonal variability in the ratio between algae and detritus.

The models also show remarkable differences in the way how the sediment-water exchange is treated. Each of the models accounts for settling of detritus material. In BALSECT this is modelled as two counteracting processes: settling and resuspension. Resuspension is gouverned by wind action. This fast dynamics was ignored in BEM and SIMBAL where only net settling is considered. Consequently, the apparent settling velocity assumed in these models must be much lower than in BALSECT. Exchange of dissolved phosphorus is incorporated in each of the models, and is modelled as a diffusive or dynamic adsorptive process. A difference, however, is whether the sediment dissolved inorganic phosphorus is assumed as constant, and thus forms just a parameter in the model (BALSECT, SIMBAL) or whether it is modelled dynamically (BEM, some versions of BALSECT). Dynamic modelling is, of course, preferable in principle, because long term effects of loading changes can only be predicted with a proper book-keeping of the sediment phosphorus house-hold. However, the present state of knowledge about the sediments makes it almost impossible to adhere sufficient confidence to the results.

The differences outlined above are structural differences. On top of these also variants exist with respect to the mathematical treatment of the various transformation processes. A detailed description would go beyond the scope of this paper; a review is presented by Van Straten and Somlyódy (1980). The most remarkable difference is observed in the description of phytoplankton mortality. In both SIMBAL and BEM mortality is modelled simply as first order in algae. In contrast, BALSECT assumes that the mortality rate is proportional to the ratio of biomass and net specific growth rate. Thus, mortality becomes a fairly complicated second order process in phytoplankton, where a large growth rate due to favourable conditions of e.g. light and temperature leads to amplified growth because of a simultaneous relatively low mortality rate. Other differences in mathematical formulation can be found in the description of the temperature dependancy of the various rate functions. It turns out, in fact, that temperature functions have a decisive influence upon the dynamic behaviour of the models. At the same time very little experimental information is available about these temperature dependancies, which constitutes an essential weakness of each of the models. Table 1 summarizes the number of state variables for the three models, as well as the number of parameters that must be specified. The demanding role of the temperature functions is very apparent from this table.

MODEL PERFORMANCE

Despite considerable progress in modeling eutrophication very few examples exist of a full model calibration and validation against an independant data set. Apart from models being necessarily strong simplifications of reality, another reason certainly also lies in the quality of available field data, due to both analytical problems

TABLE 1

	BEM	BALSECT	SIMBAL
Water Body State Variables	6 + 1 b)	5	4
Parameters			
Basic Rate Constants	13	11	8
Nutrient Limitation Factors	3 + 3 b)	1	2
Light Limitation factors a)	5	3	4
Temperature Coefficients	12	12	8
Sediment State Variables	2	2	(1) c)
Parameters			
Coefficients	4	2	3
Temperature coefficients	1 + 1 b)	-	1

Notes: a) including self-shading efficients and base extinction
b) for BEM model, + indicates extra needs for inclusion of nitrogen
c) considered as constant

as well as sampling problems. In the case of Laka Balaton the situation is similar. Although each of the three models has been calibrated more or less for one particular year, still quite strong deviations with reality can be observed. Within the scope of this paper it is not possible to present the results of the three calibrations in a comparable form. Nevertheless, some idea can be obtained from figure 8, showing the phytoplankton time pattern predicted for 1977 by SIMBAL and BALSECT in each of the four basins. The plot for BALSECT represents monthly averages taken from tables presented by Leonov (1982). It should be noted that BALSECT was calibrated upon various phosphorus fractions, rather than phytoplankton-P, which may partly explain the off-set in average level. Also the BEM-model generally predicts lower phytoplankton levels than have been observed (Kutas and Herodek; 1982). SIMBAL was calibrated using a Monte Carlo technique, in order to represent roughly the observed behaviour. The principle aim of this procedure was not calibration but hypothesis testing, and the assumption of an adsorption-desorption process was a direct result of this exercise (Van Straten, 1980). It should be noted that the analysis of primary production data as mentioned before revealed that the maximum specific growth rate should have been considerably larger than had been assumed by the time of the calibration exercise. Consequently, to maintain the fit also mortality rate must be larger. Despite problems in the calibration each of the

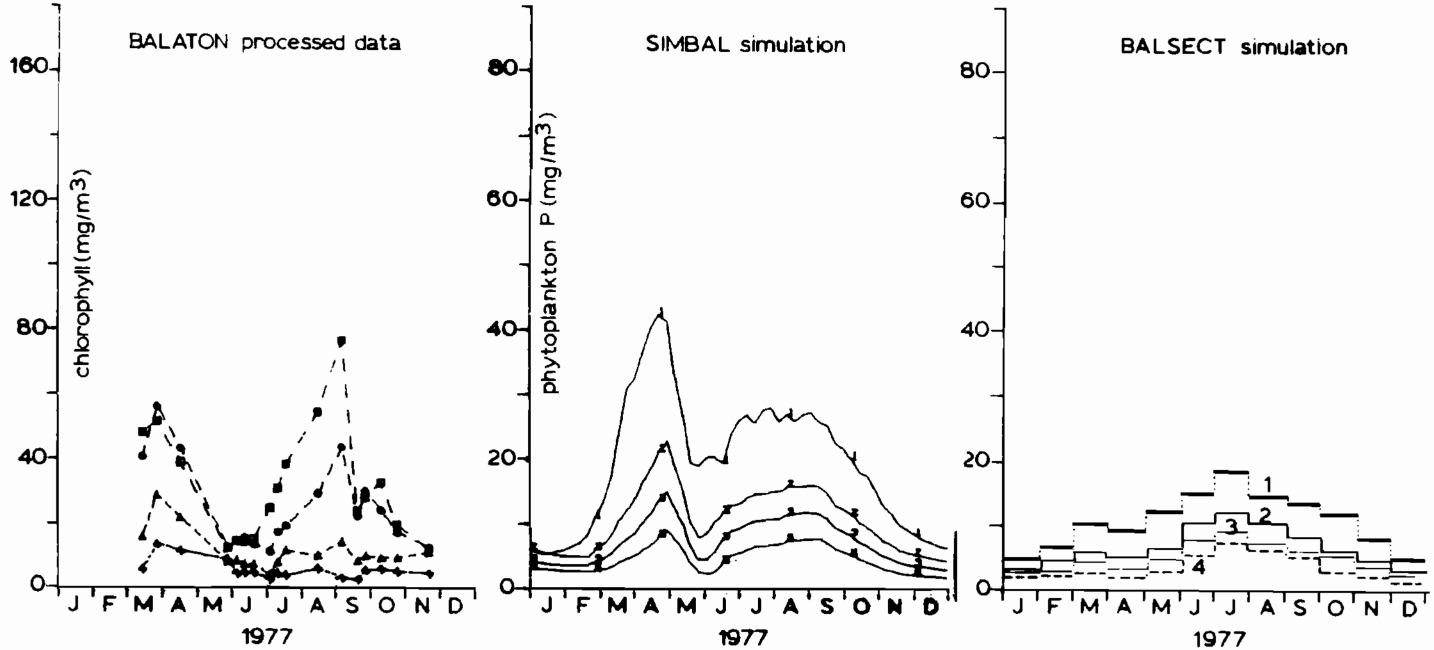


FIGURE 8: Phytoplankton-P simulation with SIMBAL and BALSECT for 1977 in each of the four basins (1=Keszthely, 2=Szigliget, 3=Szemes, 4=Tihany/Siófok) and chlorophyll-a data for comparison.

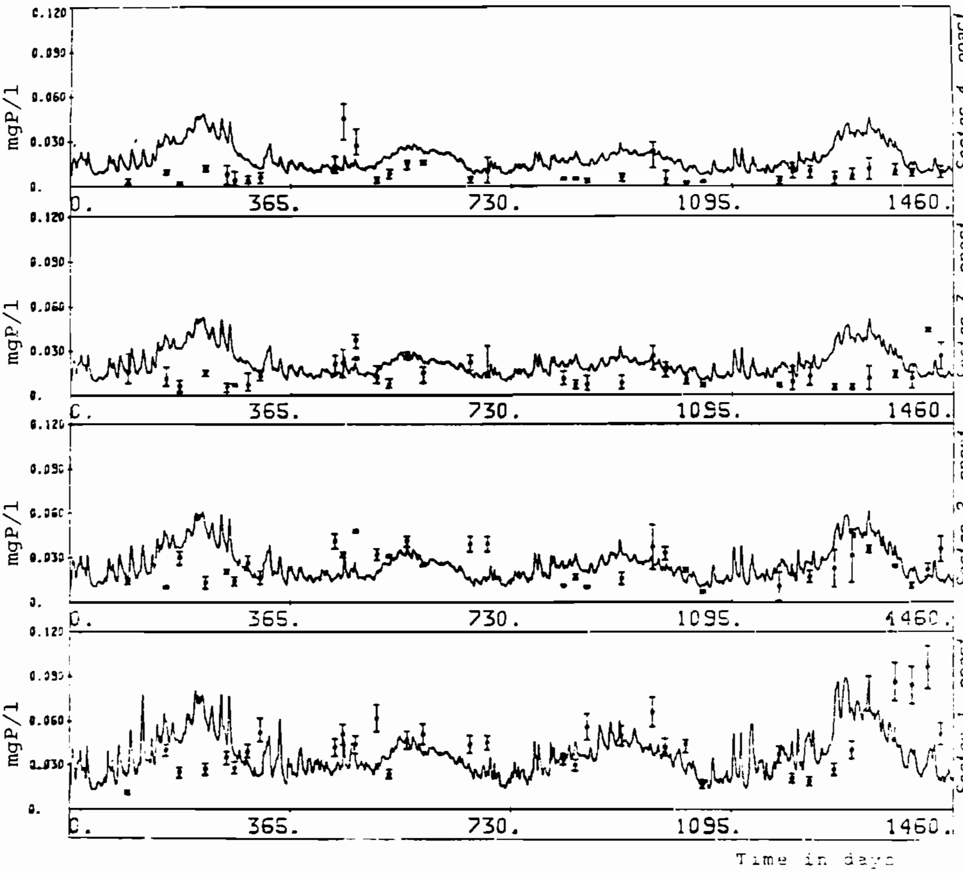


FIGURE 9: Comparison of BALSECT model calculations (curves) and observed data for the sum of phytoplankton-P, detritus-P and bacteria-P, over the period 1976-1979 (Leonov, 1982).

models was run for a sequence of years, maintaining the calibration parameter set. Figure 9 gives an impression of this for particulate organic P (i.e. the sum of phytoplankton-P, detritus-P and bacteria-P) in the BALSECT model. Again, quite strong deviations occur, but the model at least follows the trends and does not go wild. Similar results are obtained with BEM and SIMBAL.

Can we use such apparently deficient models for management? Of course, a hazard exist in doing so, because lack of fit usually means lack of understanding. However, since this is the best that can be done with the data material it may be better to use the models to have a look at the predicted tendancies, rather than using no model at all. For this purpose an investigation to see how at least the models react to changings in phosphorus loading (the most direct and efficient management option) is worthwhile. Figure 10 shows the results for both SIMBAL and BALSECT. For SIMBAL the annual average phytoplankton-P as a function of volumetric 'available'-P load for each of the basins is given (Somlyódy, 1982). The peak level is usually some 50 to 100% larger. The BALSECT plot was derived from tables presented by Leonov (1982). Rather than 'available' load the total load is considered here, but the scale has been adapted to facilitate a comparison (about half of the total load is 'available'). The reaction of phytoplankton peak to load reduction would lead to lower phytoplankton levels, although BALSECT is less optimistic than SIMBAL. It should be kept in mind, however, that BALSECT underestimates the phytoplankton levels on a whole, due to the calibration procedure as outlined before.

Whatever the situation, the load reduction behaviour according to both models shows remarkable similarities. First of all, despite quite strong non-linearities in the models the response curve is almost linear. This suggests a fairly simple relationship for management purposes, and this behaviour is also in line with the empirical predictions based upon the OECD study (figure 5). The second eye-marker, and this in contrast to the OECD results, is that both models seem to agree that a full suppression of algal blooms is not achievable, not even at zero loadings, if this were ever possible.

In conclusion one may say that dynamic models are useful and interesting tools to study the systems complexity. However, large uncertainties still exist, whereas on the other hand quite simple results are obtained for use in management applications. This suggests that simplifications should be possible for management oriented purposes, where not all dynamic detail is needed. This quite naturally leads to the concept of 'extremum analysis' as presented in the next chapter.

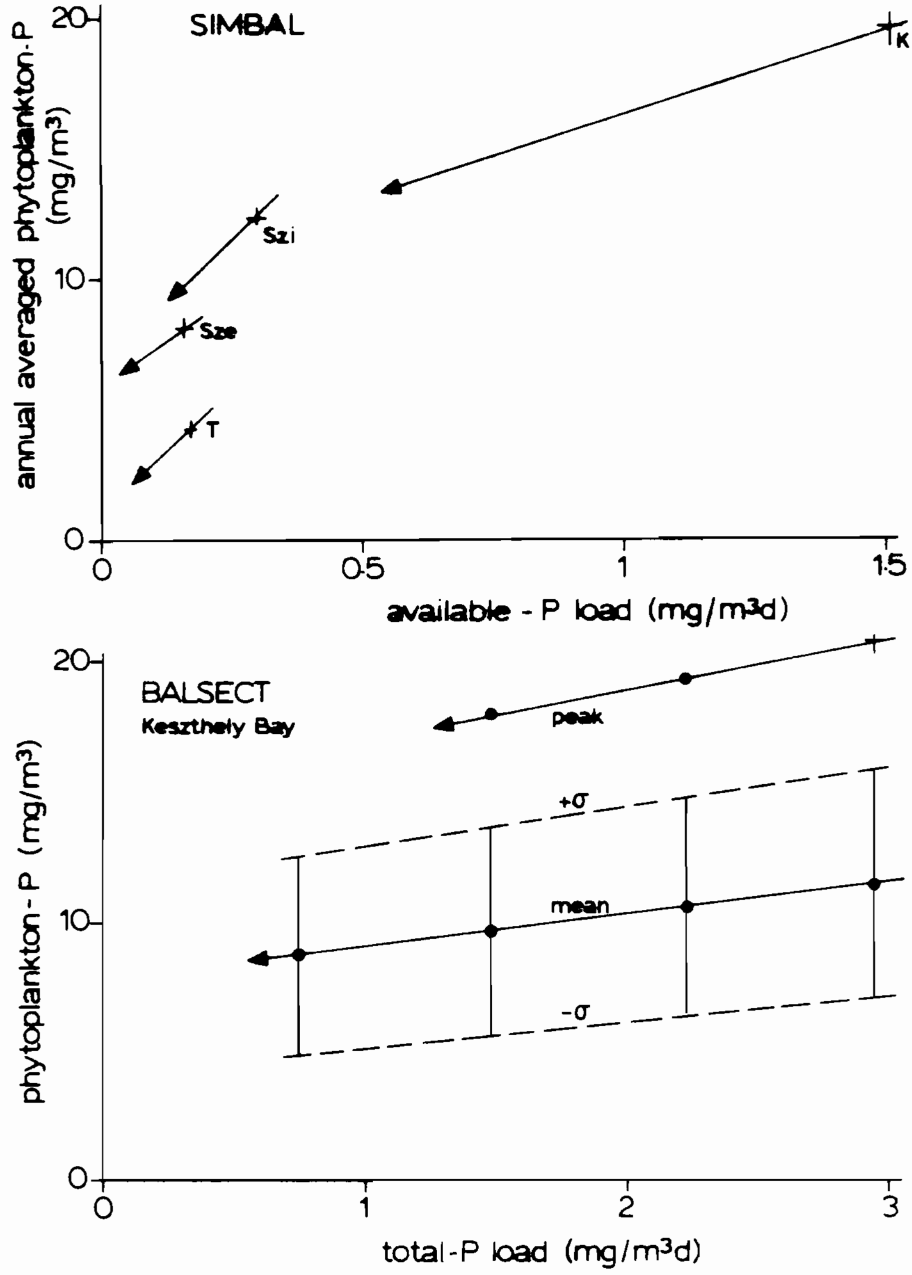


FIGURE 10: Prediction of phytoplankton-P response to phosphorus load reduction according to SIMBAL (all basins) and BALSECT (Keszthely Bay). 1977 load level for Keszthely about 3 mg $m^{-3}d^{-1}$ total P or 1.5 mg $m^{-3}d^{-1}$ 'available' P.

EXTREMUM ANALYSIS METHOD

Rather than calculating the full dynamic model in time, in this method only the extremum is looked for. This is usually what is of most interest for management purposes. To illustrate the method consider the following simplified structure:

$$\frac{dA}{dt} = k_1 f_I \frac{P}{P_K + P} A - k_2 A - qA \qquad (3a)$$

$$\frac{dD}{dt} = k_2 A - k_3 D - k_4 D - qD + l_d \qquad (3b)$$

$$\frac{dP}{dt} = -k_1 f_I \frac{P}{P_K + P} A + k_3 D - xPf_I \frac{P}{P_K + P} A + l_{int} + l_p \qquad (3c)$$

where A = phytoplankton-P

D = detritus-P

P = dissolved inorganic P

k_1 = maximum algal specific growth rate

k_2 = algal mortality rate

k_3 = detritus hydrolysis and mineralisation rate

k_4 = detritus net settling rate

f_I = light attenuation factor for algal growth, depth and day averaged

$= G(I)/(k_o + \alpha A)H \qquad (3d)$

G(I): function depending upon optimal light intensity for growth, daylength and global radiation (cf. Van Straten, 1979)

k_o : own extinction of water, including suspended solids, excluding algae

α : self-shading coefficient

H : depth

pK = Monod coefficient for phosphate limitation

q = flow rate/volume = reciproque residence time

x = biogenic lime coprecipitation factor

l_d = detritus-P loading (loading of particulate P)

l_p = dissolved inorganic P loading

l_{int} = internal loading (e.g. sediment-water exchange, adsorption/desorption).

An analysis of more complete model structures is possible, but will be presented elsewhere (Van Straten, in preparation). Now, an extremum of phytoplankton is reached if the time derivative is zero, i.e. an extremum condition is:

$$k_1 f_I \frac{P^*}{P_k + P^*} = k_2 + q \tag{4}$$

where P^* denotes the dissolved inorganic P level at the algal extreme. Since f_I depends upon the algal level, further analysis requires the solution of a set of simultaneous equations, resulting from the setting to zero of the other time derivatives (it is easy to show that this must be the case in the algal extreme). However, two situations exist for which a simplified solution is possible:

(a) the lake is not phosphorus limited. In this case $P^*/(P_k + P^*) \simeq 1$ and A is obtained by solving equation (4) directly, using (3d)

(b) the lake is phosphorus limited and turbid, i.e. $\alpha A \ll k_o$ in equation (3d). In this case f_I is not a function of A. Equation (4) leads now to the important result

$$P^* = gP_k \tag{5a}$$

with

$$g = \frac{k_2 + q}{k_1 f_I - (k_2 + q)} \tag{5b}$$

Thus, in phosphorus-limited turbid lakes the dissolved inorganic P level is directly proportional to the Monod constant, with a proportionality factor given by the ratio of loss by mortality and flushing, and nett growth. The phosphorus level is independent of the loading: any additional loading will be taken up by algae and distributed over algae and detritus such that in the end condition (4) is fulfilled again. The result presented by equation (5) also explains why the ortho-phosphate level is so remarkably constant in Lake Balaton.

Equation (3b) next provides an expression of D at the extremum algal level D^* as a function of A^*:

$$D^* = \frac{k_2}{k_3 + k_4 + q} A^* + \frac{l_d}{k_3 + k_4 + q} \tag{6}$$

Equation (6) states that the detritus level is linearly increasing with particulate P load, whereas detritus-P is coupled to algal-P by the ratio of algal mortality and total detritus loss rate.

Finally, both equation (5) and (6) can be substituted into equation (3c) to obtain the extremum algal P level:

$$A^* = \frac{\frac{k_3}{k_3 + k_4 + q} l_d + l_p + l_{int}}{(k_2 + q)(1 + xgP) - \frac{k_3 k_2}{k_3 + k_4 + q}} \tag{7}$$

In order to elucidate the significance of equation (7) a further simplification is made by setting $q = 0$. This is not unreasonable for most lakes, e.g. for Keszthely Bay $q = 0.01\ d^{-1}$ which is at least one order of magnitude less than k_2 and $k_3 + k_4$. Thus, ignoring q in (7) and introducing the short-hand notation

$$\sigma = \frac{k_4}{k_3 + k_4} \tag{8}$$

we finally obtain

$$A^* = \frac{(1 - \sigma) l_d + l_p + l_{int}}{k_2 (xgP_k + \sigma)} \tag{9}$$

This important result states that for nutrient-limited, turbid lakes the maximum algal level is practically proportional to the total P load, including both external and internal loadings. The slope of the loading-algal peak line is gouverned by only a few parameters: algal mortality rate, and the sum of losses due to biogenic lime coprecipitation and settling, the latter expressed as the fraction of the total detritus loss rate (settling plus mineralization) which is due to settling alone (σ). Thus, equation (9) provides an elegant explanation for the remarkable linearity of the model's loading responses.

In addition, now, it is clear why the predicted algal level does not drop to zero if the external load ($l_d + l_p$) is totally removed: there remains the internal load, which is, from a model point of view just a loading as any other load.

Equation (9) now also presents a nice possibility to investigate the effect of internal load uncertainty. To this end consider Figure 11. The solid line represents the SIMBAL result for the algal maximum (similar to Figure 10). Apparently, at the calibrated values for mortality rate and settling rate a considerable internal loading is implied; in fact, the internal loading should be of the same order of magnitude as the external one! The other lines indicate the effect when the internal load is different, assuming the same 'present situation' calibration point for the lake-water. Note that this implies a different mortality rate and/or settling/mineralization rate. As can be seen from the plot quite large internal load variations lead to fairly limited ranges in maximum algal level predictions. Thus, the sensitivity of the immediate algal level response to uncertainty in internal load is not as large as one would have expected.

In the development discussed thus far, two questions remain that require some consideration. First one may ask whether a steady state is ever really reached in practice. It is possible to study this problem by perturbation analysis (cf. Van Straten, 1981). In general, for Lake Balaton, loading response times are in the order of a couple of weeks to more than a month. This time period is so long that considerable changes in environmental factors such as temperature and light can be expected, which in turn influence the coefficients in equations (4) through (9). Consequently, equilibrium will not be reached at the time of optimal light and temperature conditions; rather, the algal peak lags behind and will be reached when light and temperature have already started declining again. Nevertheless, the extremum analysis remains valid, formally, but is more difficult to use because the choice of the proper coefficients at the time of the extremum is more uncertain. In any case, using the optimal conditions will lead to an overestimation of algal peak levels, and the calculated levels can therefore be viewed as an upper bound.

The second question is whether a load reduction will not affect the internal load. Of course, such a question can only be answered when appropriate knowledge about the sediment is available. Here we are interested in investigating the implications of an (assumed) feed-back mechanism within the sediment. The starting point is Figure 11 which holds if no feedback occurs. Now, let us assume a relation between external load and internal load. Such a hypothetical relationship (more or less in the form of an adsorption isotherm) is depicted in the right-bottom half of Figure 12. Then, any external load reduction entails a change in internal load (which may, however, be delayed. These dynamics are ignored here). If this new internal load is projected upon the internal load axis, a new lake response line may be drawn. As long as algal mortality rate and mineralization and settling are not affected this line will be parallel to the original response. The new algal peak is now lower than originally, due to the effect of the feed-back. Finally, when the external loading goes down to zero, the algal peaks drop to almost zero too. This result is more in agreement with the empirical findings from the OECD study.

These results are very important for management. In the light of Figure 12 the loading response of the dynamic models may be viewed as the immediate response, i.e. the response which can be expected in the year of implementation of a loading reduction measure. It should be noted, however, that this response may be masked considerably by year-to-year fluctuations in meteorological factors (cf. Somlyódy, 1981). The immediate response is not very large and may discourage the implementation of abatement programs. However, Figure 12 suggests that this is not a correct conclusion: due to the feed-back mechanism within the sediment the internal load will also drop, and sooner or later algal peak levels will go down further. Thus, the perspectives for management are better than suggested by the present version of the dynamic models. The results also indicate that inclusion of long-term sediment dynamics in the models is absolutely necessary for the achievement of better long-term predictions. From this point of view the research effort presently put on the better understanding of sediment processes is fully justified.

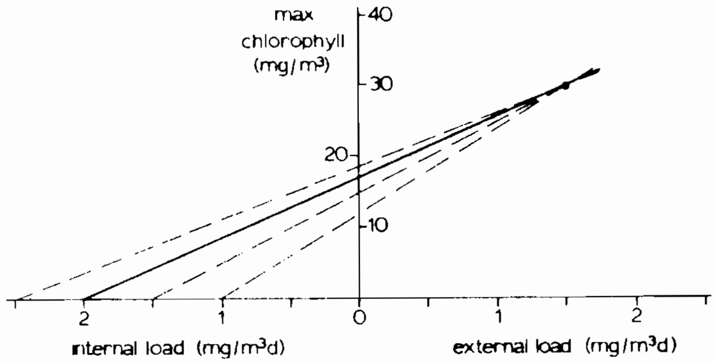


FIGURE 11: Effect of internal load uncertainty upon phytoplankton-load relationship. Read the immediately attainable phytoplankton levels for zero external load from where the lines hit the chlorophyll axis.

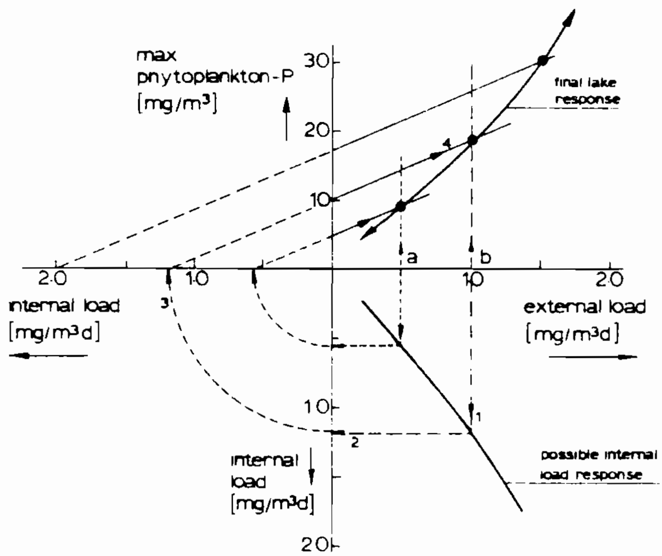


FIGURE 12: Ultimate phytoplankton response to external load reduction with (hypothetical) feed-back upon internal load. For any external load read the new equilibrium internal load from the bottom-right sector (point 1-2), project upon horizontal internal load axis (point 3) and draw the new immediate response line parallel to original one. The new response is at the intersection of this line with the external load (point 4).

CONCLUSIONS

With special emphasis on managing the eutrophication problem the following conclusions emerge from the discussion in this review report:

(i) empirical relationships such as the Vollenweide/OECD plots are useful to underline the close relationship between external phosphorus loadings and algal blooms. However, the accuracy for application to a specific lake is not always sufficient.

(ii) dynamic modelling provides a powerful tool, in principle. The development of models has led to a significant gain in understanding and has served as a guide to direct further scientific research. However, in their present state, models are not yet fully adequate, for example because of lack of appropriate sediment dynamics.

(iii) simple load-peak relationships by static analysis of dynamic models have prooved helpful to detect the shortcomings of these models. The sediment behaviour turns out to determine the transient lake response between the present and the future load situation. But in any case, on the long term, phosphate load reduction is more beneficial than expected.

(iv) in the light of the conclusions above, management attention should be concentrated upon the identification of the most cost-effective phosphorus load reduction method. In the case of Lake Balaton reduction of sewage loads is most likely the first thing to do.

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Appendix A: Calculation of the OECD/Vollenweider eutrophication model parameters for the four basins of Lake Balaton

Quantity	Units	Symbol	Kes	Szi	Sze	Tih	Balaton total	Reference
Tributory inflow	m/y		8.08	1.28	0.32	0.12	0.97	(1) p. 18
Throughflow in	m/y	I_i	-	2.06	2.36	1.94	-	(1) p. 18
Precipitation	m/y		0.63	0.63	0.63	0.63	0.63	(1) p. 18
Total	*m/y*	I	*8.71*	*3.97*	*3.31*	*2.69*	*1.60*	
Surface area	$10^6\ m^2$	A	38	144	186	228	596	(1) p. 16
Inflow	$10^6\ m^3/y$	Q	331	572	616	613	954	Q=A.I
Volume	$10^6\ m^3$	V	82	413	600	802	1907	(1) p. 16
Residence time	y	τ_w	0.25	0.72	0.97	1.31	2.00	$\tau_w=V/Q$
Mean depth	m	$\bar{z}$	2.16	2.87	3.23	3.52	3.20	$\bar{z}=V/A$
Hydraulic load	m/y	q_s	8.7	4.0	3.3	2.7	1.6	$q_s=\bar{z}\tau_w$
P load total	kg/d	L_E^*	262	273	227	253	1015	(2) p. 36
P load total	kg/y	L_E	95360	99645	82855	92345	370475	$L_E^* \cdot 365$
Average TP concentration	mg/m^3	P	78	52	35	28	-	'76,'77,'78 data
Throughflow load	kg/y	L_Q	-	23140	22825	15480	-	$L_Q=P_{i-1} \cdot I_i A_i$
Total load	kg/y	L_T	95360	122785	105680	107825	370475	$L_T=L_E+L_Q$
Areal load	$mg\ P/m^2\ y$	L	2517	853	563	473	622	$L=L_T/A$
'Normalized' Areal Load	$mg\ P/m^3$	$P_{\lambda L}$	193	116	86	62		$P_{\lambda L}=\frac{L}{q_s}\frac{1}{1+\sqrt{\tau_w}}$
Average TP concentration	mg/m^3	P	75-81	46-63	31-42	24-30		
Max. Chlorophyll-a	mg/m^3		40-70	30-60	25-30	15		

See figures 1 and 2

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LAKE EUTROPHICATION MANAGEMENT MODELS

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1. INTRODUCTION

Mathematical models can serve as useful tools for eutrophication management. In general, two kinds of model applications can be observed in the literature:

/i/ Simulation models of lake eutrophication /or empirical load response relationships/ are frequently employed to determine the load reduction required to meet water quality goals. The questions, how to achieve this reduction and what economic consequences should be accounted for, are rarely analyzed in details.

/ii/ The other group of models disregards the in-lake processes, e.g. simply aiming at minimizing external loads by the available control alternatives, under budgetary constraints. The final management model is based on an optimization procedure comprising the nutrient loading model.

Evidently, the combination of simulation and optimization approaches should be more powerful than solely one of them. In most cases the objective of management is formulated through the intended lake water quality /e.g. the water body should be shifted from hypertrophic to eutrophic state/

regarding the actual economic constraints, too.
For this reason, one of the objectives here is to develop an optimization management model incorporating properly the dynamic submodels or knowledges disseminated from them thus allowing to make "macroscopic" decisions based on sound, "microscopic" studies on various subprocesses. In other words, the ultimate goal is to establish the water quality management model on the highest stratum of the present analysis /as discussed and shown by Somlyódy in this proceedings/ having the scientific bases /and gaps, too/ as summarized in the preceding General Reports. Through such a development several important questions can be answered. For instance, how sensitive is the model on various watershed-, and in-lake processes? How do the uncertainties propagate and appear on the level of decision making?

A management model of this fashion should involve also the influence of some subjective factors deriving from the nature of the policy making procedure in the frame of which such models are used. Again, the question is, how sensitive is the model performance on these factors especially as contrasted to the influence of scientific subprocesses and uncertainties?

The study outlined below was directed not only for solving seemingly methodological questions. Inversely, the objective was to contribute to the solution of the particular water quality problem of Lake Balaton, meanwhile several methodological questions quite general in character raised. Several alternative management models were formulated /as was also done for hydrodynamics or other in-lake processes/. Again the intercomparison of conclusions gained from various models is of interest.

This report is organized as follows. Section 2 outlines the structure of analysis which comes from the principle of decomposition and aggregation. Subsequently, an aggregated, deterministic and stochastic linear load response model is derived. The procedure is based on computer experiments with a dynamic lake eutrophication model. Section 4 presents the management model. The stochastic problem is simplified by introducing a proper deterministic objective function and solved by linear programming. Results on the short-term control strategy for Lake Balaton are given in Section 5. Subsequently four other models and their conclusions are discussed briefly, and are compared to the findings of the present approach. The paper is completed by conclusions.

2. STRUCTURE OF THE ANALYSIS

The procedure adapted is illustrated on Fig. 1. /see also Somlyódy in this proceedings, Fig.5/. Dynamic lake eutrophication model, LEM 1 has both controllable and non-controllable inputs, the mainly artificial nutrient loadings and meteorologic factors, resp. Both of them are derived from field observations when developing LEM 1, but in the planning phase some "critical" scenarios should serve as a basis or the inputs are to be generated in a random fashion. It is noted, that the segmentation of the nutrient

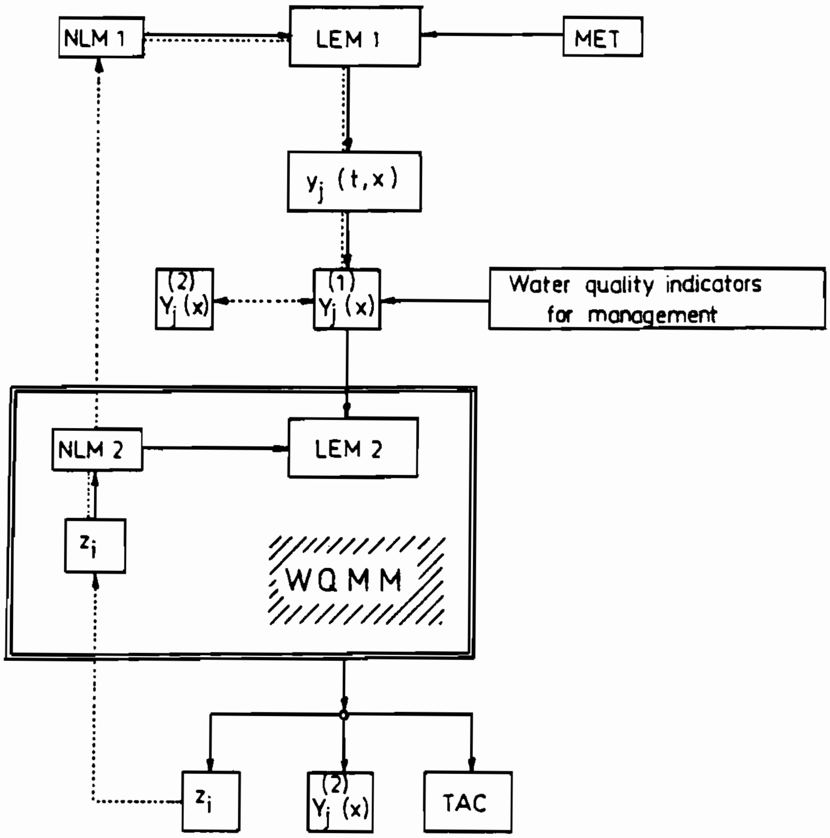

Fig. 1. Structure of the analysis

LEM - Lake Eutrophication Model;
NLM - Nutrient Loading Model;
WQMM - Water Quality Management Model;
MET - Meteorologic Forcing Functions;
y_j - state variables of LEM,
Y_j - water quality indicators
z_i - control variables;
TAC - Total Annual Cost

loading model NLM 1, corresponds to that of LEM 1, i.e. the total load reaching a basin of the lake is of interest rather than the particular loads from and location of the individual sources on the connected subwatershed.

The model output results in the spatial and temporal changes of state variables /e.g. various phosphorus components/, $y_j(t,x)$, characterizing the water quality. These changes should be known on the level of understanding, but for decision making much simpler indicators reflecting the global behaviour of the system can be used. For Lake Balaton, perhaps the only suitable parameter is the chlorophyll-a concentration frequently employed in the literature for defining the trophic state classes of water bodies. The chlorophyll content affects the colour of water and thus it is a proper measure of water quality, i.e. of the recreational value of the lake.

Eventually the yearly peak chlorophyll-a concentration was selected as a single indicator: taking into account the cumulative character of the eutrophication process and the relatively slow response to control measures /equal to the flushing time in order of magnitude/ the annual dynamics is of secondary importance. The situation is similar for many other lakes since in general a relatively close correlation can be observed between the yearly average and maximal Chl-a concentrations /OECD, 1982/.

The selection of $Y = /\text{Chl-a}/_{max}$ as a single indicator for management has one major consequence, namely time can be deleted and the dynamic lake model used in an off-line fashion for the subsequent optimization. Various possibilities exist in this respect. For example, LEM 1 can be run for a large variety of realistic forcing function scenarios and then $Y_j(x)$ values are stored on computer. These values represent a surface on which the solution for the management problem is then looked for. The other possibility followed here is to parametrize properly the results of LEM 1 thus deriving an aggregated model called LEM 2 which e.g. describes approximately the indicator as a function of the nutrient load. Based on available experiences /see e.g. empirical models/ it is hoped that LEM 2 has a simple algebric structure easy to handle later on.

LEM 2 is interconnected with a nutrient loading model, NLM 2 including the control variables, z_i. As compared to NLM 1, the model used here contains more details for the watershed, but it is aggregated as the time is concerned /see later/.

The coupled NLM 2 - LEM 2 models are then replaced in an optimization procedure which - depending on the formulation of the water quality management model /WQMM/ - e.g. results in the corresponding values of control variables, z_i, indicators, $Y_j^{(2)}(x)$ and total annual costs, TAC /see Fig.1/.

As a final step of the analysis, the detailed lake eutrophication model, LEM 1, can be run with the "optimal"[1] input scenario and the accuracy of LEM 2 checked as indicated on Fig.1.

Even if LEM 1 has a deterministic structure - which is

1/ Feasible or quasy-optimal solution is meant here rather than optimum in a strict mathematical sense.

the case here - the forcing functions can be both deterministic and stochastic in character. Accordingly, the same is true for the model components NLM 2, LEM 2 and WQMM. The decision in this respect - whether to follow a deterministic or stochastic formulation - depends mainly on the character of the problem and the analyst itself, as well.

In the subsequent sections the procedure outlined above is discussed step by step for Lake Balaton.

3. DERIVATION OF AGGREGATED MODELS

At this stage of the study it is assumed that a dynamic eutrophication model is already available. Of the models discussed by van Straten /this proceedings/, SIMBAL was selected,but all the important conclusions gained were tested also by other models.

3.1. Input data averaging

The computer experiments revealed that the model is not sensitive to day-to-day changes in forcing variables and monthly means can be satisfactority used for loading, temperature and global radiation. This is an important practical finding as the generation of forcing functions for shorter averaging period would be hampered by lack of data.

3.2. Input generation in a random fashion

For Lake Balaton we failed to find "critical" /or "design"/ environmental conditions which could have served as a basis for management /the situation should be the same for many other shallow lakes/. Consequently the model inputs were considered stochastic variables.

Nutrient loadings

For generating monthly phosphorus loads, $\bar{L}$, the daily observations of Joó /1980/ on the Zala River for the period 1976-1979 and 25 years long records for streamflow rate, Q /Rákóczi and Varsa, 1976/ were utilized. Autocorrelative features are ruled out from time series of monthly means and a simple regression expression between $\bar{L}_i$ and $\bar{Q}_i$ was found acceptable. The statistical properties of $\bar{Q}_i$ are given by various gamma and log-normal distributions /Rákóczi and Varsa, 1976/, while the residual error of $\bar{L}_i = f(\bar{Q}_i)$ regression equation is approached by normal distribution.

Using again the Zala River data set, a Monte Carlo analysis was performed /Somlyódy, 1983/ in order to study the uncertainties caused by infrequent sampling /one or two samples per month for most of the tributaries/. The error related was described by two parameter gamma distributions fitted to the results of Monte Carlo simulations.

The loading generator developed for the River Zala covering 50 % of the Lake Balaton catchment was subsequently extended for other tributaries and the extrapolated sampling error term added. The loading model derived in this manner is fairly accurate for the most endangered region of the lake, the Keszthely bay /although the records were still not long enough/, but for the other basins it should be regarded as an approximation only, since the distributions typical for the Zala watershed had to be used due to the lack of detailed data on other catchments. Having in mind, however, that the volume and residence time of these basins are much larger and consequently the influence of the random changes on water quality is anticipated to be smaller, the model can be considered realistic for the present purpose.

As noted before, the model works on a monthly basis. In the management model, however, yearly averages can be satisfactorily used /see later/. For this reason the model was further aggregated for calculating yearly means /NLM 2, see Fig.1/.

Temperature and global radiation

Based on a ten years long historical record a simple autoregressive model was developed to calculate daily water temperature /the length of ice covered period is generated independently/. While temperature shows a moderate change from day-to-day, global radiation is characterized by large fluctuations. For this reason radiation data were "a priori" aggregated to weekly averages. Since cross-correlation is important here, radiation was related to weekly mean temperatures and eventually a model incorporating the temperature at two subsequent weeks was arrived at. Finally the daily temperature and weekly global radiation is aggregated to monthly means for the input of lake eutrophication model.

3.3. Use of the lake eutrophication model under stochastic input

Shallow lakes are strongly affected by random fluctuations in environmental factors and the quantification of this influence is of major importance for proper management. For this reason, input generators outlined in section 3.2. were coupled to LEM which was then used in a Monte Carlo fashion.

Fig.2. shows the influence of climatic, non-controllable factors on chlorophyll-a concentration /a conversion factor of 2 from phytoplankton P /PPP/ to Chl-a was adapted; see van Straten in this proceedings/. As compared to the simulation results gained with SIMBAL for 1977 /van Straten, this proceedings, Fig.8/ the changes in the average trajectory are less pronounced. Namely, the mean of Monte Carlo simulations is determined primarily by the average input scenarios which in fact can hardly occur in nature. It is apparent from the figure that the annual peak chlorophyll-a concentration, $/\mathrm{Chl\text{-}a}/_{max}$ can range between 30 and 90 mg/m^3 /$\pm$ 40% around the mean/ depending solely on climatic factors; a strikingly wide domain. Such fluctuations can seemingly compensate the effect of considerable load reductions. The sensitivity of other basins of the lake is smaller: the coefficient of variation of $/\mathrm{Chl\text{-}a}/_{max}$ for Basin II...IV is 10, 6 and 5% /it is 13% for Basin I/. The upper extremes are 50, 30 and 20 mg/m^3 in the same sequence.

If stochastic changes in loads are accounted for, too, Fig.3. is resulted. The coefficient of variation is now about

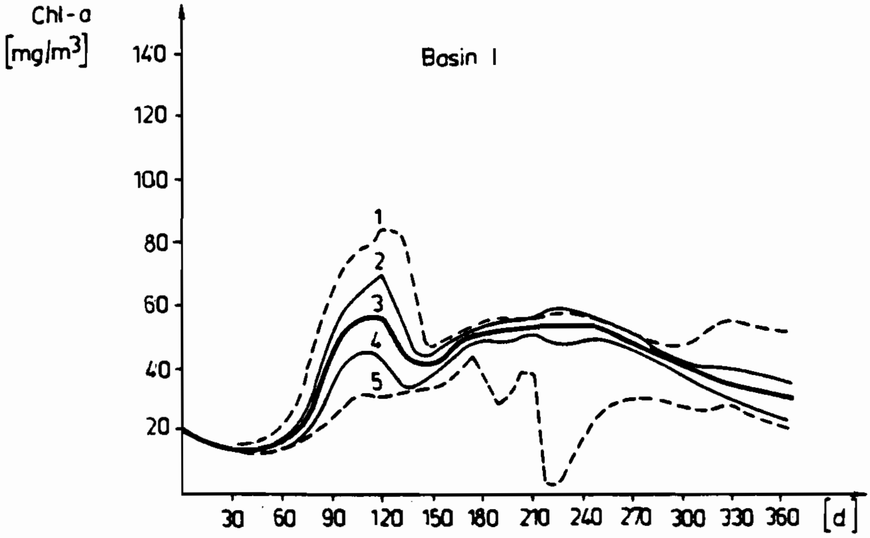

Fig. 2. Influence of non-controllable factors on water quality

3 - mean; 2 and 4 - ± standard deviation;
1 and 5 extremes /all from 200 Monte Carlo simulations

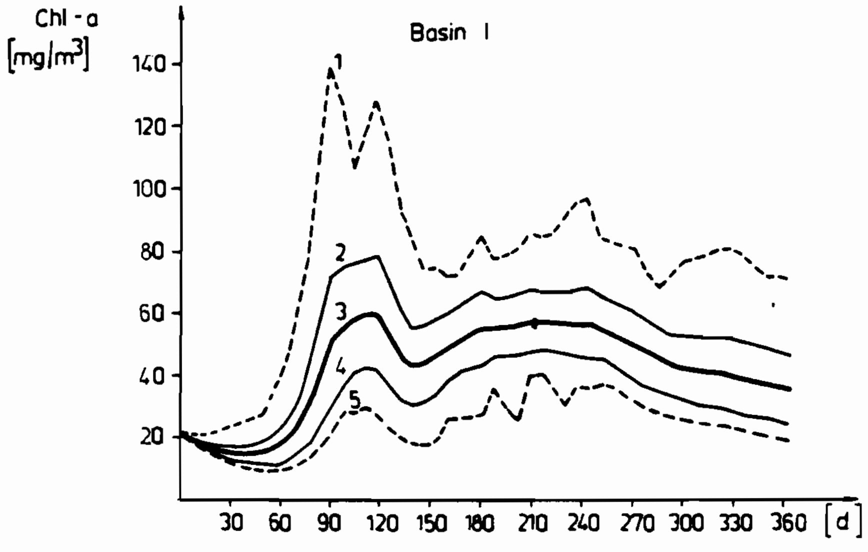

Fig. 3. Influence of natural and controllable factors on water quality

3 - mean; 2 and 4 - ± standard deviation;
1 and 5 extremes /all from 200 Monte Carlo simulations

20 % and the extreme value of $/Chl\text{-}a/_{max}$ can reach 140-150 mg/m^3. Upper extremes for the subsequent basins are 60, 35 and 25 mg/m^3, resp. While empirical distributions of $/Chl\text{-}a/_{max}$ were approximately simmetrical in the first case /Fig.2/, here typically skewed distributions are resulted which is also reflected by Fig.3.

3.4. Aggregated lake eutrophication model: deterministic version

Dynamic lake eutrophication model, SIMBAL, was run under systematically changed load conditions. Subsequently $/Chl\text{-}a/_{max}$ was plotted against the yearly average values of biologically available phosphorus load, L_{BAP}, and the linear relationship as shown on Fig.4 was gained. The linearity is surprising for the first glance only: the same trend is reflected also by empirical models /OECD 1982/. It is perhaps a little disappointing that the practical essence of sophisticated dynamic models is so simple, however as the extremum analysis by van Straten /this proceedings/ shows, the linear load response may be valid quite widely for nutrient limited, turbid lakes /at least on the present level of our understanding/. It is noted that all the other models developed for Lake Balaton have this feature /see also Baker and Harleman in this volume/. Similar conclusions were arrived at for Lake Erie, too /see Lam and Somlyódy in this proceedings/.

Apparent from Fig.4. that $/Chl\text{-}a/_{max}$ is related linearly to the sum of external and internal loadings /L_{BAP} and L_{IP}, resp./ and thus $/Chl\text{-}a/_{max}$ does not necessarily approaches to

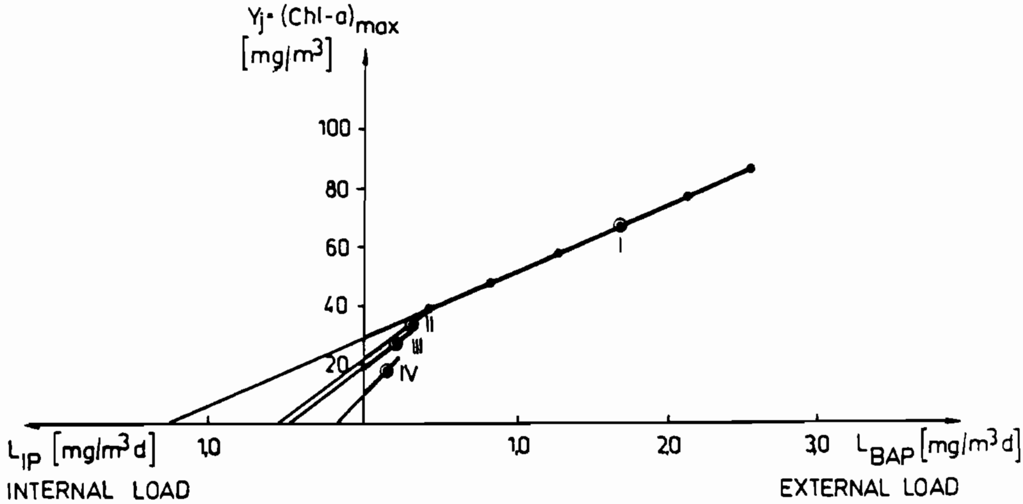

Fig. 4. Aggregated lake eutrophication model: deterministic version

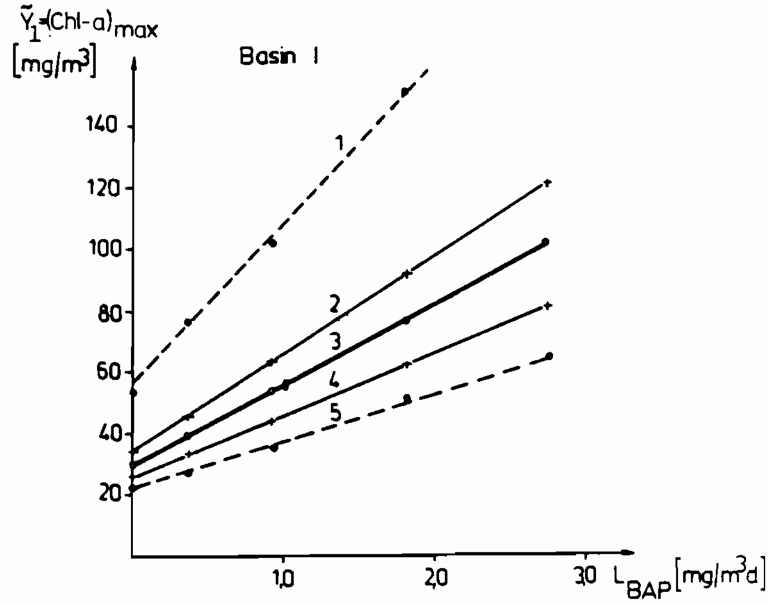

Fig. 5. Aggregated lake eutrophication model; stochastic version
3 - mean; 2 and 4 - $\pm$ standard deviation; 1 and 5 extremes

zero /or a relatively small value/ if $L_{BAP} \rightarrow 0$. From Fig.4. the yearly average internal load can be estimated to about 6-800 kg/d / $\sim$ 1.2 mg/m^{3}d/ which is of the same order of magnitude than the external load. This value is in harmony with the estimate given by Lijklema et al on the basis of experimental results /see in this proceedings/.

The particular model, SIMBAL, used here incorporates only implicitly the influence of sediment, but the long term behaviour of sediment /enrichment and renewal/ is not modelled[1] due to the lack of existing information and experiences /see Lijklema et al in this proceedings/. Consequently, Fig.4. does not give information on the progress of eutrophication or how future equilibrium is reached. The plot illustrates the short term /deterministic/ response of the lake. As the external and internal loads are coupled /the reduction of the external load generates a decrease in the internal load with some lag, see Lijklema et al. and van Straten in this proceedings/ the long term improvement is larger than suggested by Fig.4. /e.g. according to the plot Basin I would remain eutrophic even if external load were cut completely/.

The important conclusion of Fig.4. is that the dynamic model, LEM 1, can be replaced by an aggregated linear model version, LEM 2 on this level of analysis :

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1 Formally the inclusion of the "memory effect" of sediment is easy to accomplish /see Kutas and Herodek in this volume/ but calibration can be based solely on intuition.

$$\underline{Y} = \underline{Y_O} + \underline{\underline{A}} \, /\underline{L}_{BAP} - \underline{L}_{BAPO}/ = \underline{Y_O} - \underline{\underline{A}} \, \Delta \underline{L}_{BAP} \qquad /1/$$

Here $\underline{Y}$ and $\underline{L}_{BAP}$ vectors represent $/Chl\text{-}a/_{max}$ $[mg/m^3]$ and volume related loadings $[mg/m^3d]$, resp. for the four basins and O indicates the present /or nominal/ state.

The elements of matrix $\underline{\underline{A}}$ are reciproques of lumped reaction rates $[d]$. The main diagonal comprises primarily the effect of biological and biochemical processes /with their forcing functions/, while other elements that of the inter-basin exchange /due to hydrologic throughflow and mixing/.

For Lake Balaton only the neighbouring basins are coupled /in a unidirectional way/ thus matrix $\underline{\underline{A}}$ takes the specific form

$$\underline{\underline{A}} = \begin{bmatrix} a_{11} & 0 & 0 & 0 \\ a_{21} & a_{22} & 0 & 0 \\ 0 & a_{32} & a_{33} & 0 \\ 0 & 0 & a_{34} & a_{44} \end{bmatrix} \qquad /2/$$

indicating that any management actions performed on the eastern subwatersheds have no effect on water quality of the western basins. Values of $a_{11} \ldots a_{44}$ range between 20 and 35, while $a_{21} \ldots a_{34}$ are smaller by an order of magnitude.

It should be mentioned that the global influence of sediment renewal can be easily included in Eq./1/. If the equilibrium relation, L_{IP} $/L_{BAP}/$ is linearized in the vicinity of the nominal point and the corresponding tangents are $k_1 \ldots k_4$ $/>0/$, the long term response can be gained formally by using a modified matrix

$$\underline{\underline{A}}^{*} = \begin{bmatrix} /1+k_1/a_{11} & 0 & 0 & 0 \\ /1+k_1/a_{21} & /1+k_2/a_{22} & 0 & 0 \\ & /1+k_2/a_{32} & /1+k_3/a_{33} & \\ 0 & 0 & /1+k_3/a_{34} & /1+k_4/a_{44} \end{bmatrix} \quad /3/$$

Again, the problem is that the k_j values are unknown and only loose statements can be given on the time required to reach equilibrium /see Lijklema et al and van Straten in this proceedings/.

Nevertheless, it is worthy to mention, that the simplified model arrived at still preserves the influence of all the important subprocesses /discussed in the preceding General Reports/ in an aggregated way.

3.5. Aggregated lake eutrophication model: stochastic version

Monte Carlo simulations outlined in Section 3.3. were also performed under various reduced load conditions. In order to separate the influence of controllable and non-controllable factors on water quality indicator, two runs were done in each cases

/i/ with climatic generator, only and

/ii/ with simultaneous generation of nutrient loadings and climatic factors.

Results for the Keszthely basin are given on Fig.5. /situation /ii//.

As it appears from the figure, linearity is held as before, and not only for the mean, but also for statistical properties. The sensitivity of the model /and lake/ is strikingly large on variations in weather conditions. Taking into account also the "slow" recovery of sediment, this means that after reducing the external loadings, the lake approaches the new equilibrium through considerable fluctuations: managers and public opinion should be retained from expecting spectacular improvements in water quality.

Carefully analyzing the results of Monte Carlo experiments an approximate separation can be done for the influence of natural and controllable factors, resp. The expression gained is as follows

$$\underline{\tilde{Y}} = \underline{Y}_o + \underline{\tilde{Y}}_w - /\underline{\underline{A}} + \underline{b}\underline{\tilde{Y}}_w/ \ \Delta\underline{\tilde{L}}_{BAP} \qquad /4/$$

where $\underline{\underline{A}}$ and $\underline{b}$ are derived from model simulations, wavy line indicates stochastic variables, and $\Delta\underline{\tilde{L}}_{BAP}$ is defined as the difference of the non-controlled deterministic load and the controlled stochastic load.

According to Eq/4/ the water quality varies randomly for three reasons:

/i/ random changes in temperature and solar radiation /elements of $\underline{\tilde{Y}}_w$ are given by gamma distributions/;

/ii/ stochastic changes and uncertainties of the loading /Section 3.2/; and

/iii/ the combined effect of climatic and loading factors.

With Eq./4/ the aggregated lake eutrophication model, LEM 2 looked for is given. It describes the annual peak chlorophyll concentration for the four basins as a function of yearly mean L_{BAP} load, accounting also for various uncertainties in the system. This model is used in the subsequent analysis.

4. FORMULATION OF THE MANAGEMENT MODEL

Large variety of objectives can be specified for a management model developed for a given situation. This will be especially apparent in Section 6. Here the objective is to work out the short term "optimal" control strategy for Lake Balaton. In a slightly different formulation, our intention is to develop a supporting tool employed in the policy making procedure for establishing short term strategy.

Long term management action plan exist for the lake which incorporates sewage diversion and pre-reservoirs in the focus, furthermore other measures such as melioration, dredging etc. When elaborating this plan around the mid seventies the progress of man-made eutrophication was thought much slower than observed nowadays. Accordingly, few large projects regional in character were intended to realize with a relatively slow scheduling determined by economic conditions.

Recent scientific results have revealed however, that the lake is now in a quite labile state. The uncertainty analysis /Figs. 3. and 5./ based on data typical for mid and late seventies has shown that chlorophyll concentration can reach 150 mg/m^3 in the Keszthely bay /a hypertrophic value/ which would mean a doubling compared to years 1977-1981 /prognosis made in 1981, Somlyódy, 1981/. In 1982, under "unfavourable" hydrometeorological conditions /high temperature, proper light conditions for algal growth and frequent rainfalls/ $/Chl\text{-}a/_{max}$ near to 200 mg/m^3 was observed /see Somlyódy in this proceedings, Fig.3/.[1]

This acceleration of artificial eutrophication obviously calls for a set of protection measures which can be realized quickly /even with a provisional character/ complementing the long term plan.

For establishing short term control strategy not too many principal alternatives are available just tertiary sewage treatment on the existing plants without developing the sewer network and pre-reservoirs at the tributaries before they enter the lake. Through these two options point-, and surface non-point sources /mainly of agricultural origin/, resp. can be controlled.

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[1] At the same time the composition of phytoplankton changed considerably, too and a mass spreading of blue-green algae took place in August. Admittedly such changes can not be predicted quantitatively at the present stage of ecological knowledges. This fact ultimately forces the analyst to transmit gradually the available and perhaps scientifically not completely accurate information to the practice in order to make decisions on the basis of this knowledge rather than on that of "no" knowledge. The danger is, if the analyst waits for "final scientific" results, that up this time a drastically different system can be find and conclusions will be unusable for practice.

Thus, the goal of the management model is to give the "optimal" distribution of investments on the watershed.

The locations of major control possibilities are given in Fig.6. together with the corresponding load data /see Jolánkai and Dávid in this volume/. These cover about 85% of the total controllable nutrient load/the rest represented by several small creeks and sewage discharges could be involved with no difficulty/.

The aggregated lake eutrophication model /describing short term responses as required here/ to be used in the management framework was already outlined. Subsequently, the control variables, cost functions, the connection of nutrient load and control variables, objective function and constraints, furthermore the solution technique adopted and evaluation of results are discussed.

4.1. Control variables

One class of control variables considered basically continuous are the phosphorus removal rates of tertiary treatment and reservoirs /$0 \leq z_i \leq 1$/. Sometimes, it is feasible to use /0,1/ variables for reservoirs or prescribe lower and/or upper bounds, $0 \leq z_{il} \leq z_i \leq z_{iu} \leq 1$ /e.g. fixing of the minimum size/.

Reservoir systems[1] are assumed to consist of two parts /see Fig.6/, one for the removal of particulate P through

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[1] Such systems may be of multipurpose use; here they are considered as tools for water quality control, only.

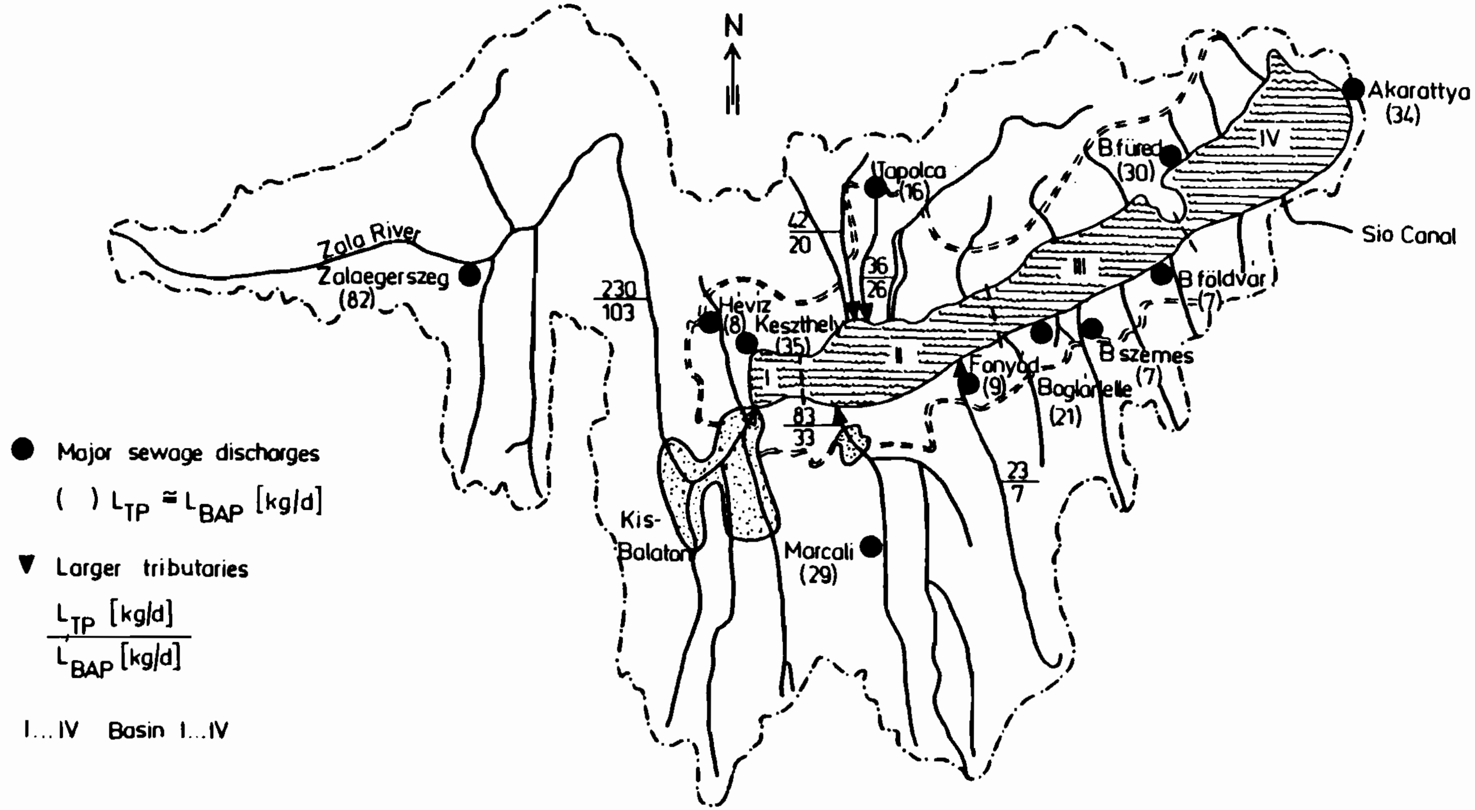

Fig. 6. Major nutrient sources and control options

sedimentation, while the other for dissolved P /benthic eutrophication in "reed-lake" elements, sorption etc./.

Lower bounds often exist also for control variables of tertiary treatment: the effluent concentration is prescribed /the standard is 2 g/m^3 in Hungary/.

The other class is formed by variables having the values of 0 or 1 all the time. With the aid of these variables and logical constraints it can be taken into account that the biological treatment should be intensified at most of the plants and investments of the upgrading has to precede the introduction of P precipitation.

4.2. Cost functions

The data on upgrading costs were taken from the report of Benedek and Szabó /1981/ offering a survey on the present stage of treatment and possible improvement for the largest plants in the Lake Balaton region. Investment-, and operational costs of tertiary treatment were elaborated on the basis of the same report and literature /e.g. Monteith et al 1980., Schüssler 1981/. The cost functions are highly non--linear depending on the effluent concentration and/or removal efficiency /costs will be doubled if 1 g/m^3 effluent concentration is required instead of 2 g/m^3, whilst about ten times larger for ≤ 0.5 g/m^3/. Obviously the cost functions are different for each sewage plant. They are linearized piece-wise for the water quality management model which requires new variables to introduce /see e.g. Loucks et al 1981/.

Investment costs of reservoirs as a function of control variables are linear as it was established from the knowledge available for the largest project at the mouth of the Zala River /Fig.6/ called Kis- /or Small/ Balaton. Running costs are neglected /or assumed to be compensated by benefits of reservoirs, e.g. utilization of reed harvested/.

4.3. Relationship between nutrient load and control variables

Let us consider the j_{th} basin of the lake which is fed - for the sake of simplicity - by a single river, furthermore by several direct and indirect sewage loads, L_{DSP} and L_{ISP}, resp. Then the change in the basin's BAP load, ΔL_{BAP}, can be expressed as

$$\Delta\tilde{L}_{BAPj} = z_2 E(\tilde{L}_{DRP}) + (z_2-1)[\tilde{L}_{DRP} - E(\tilde{L}_{DRP})] + a\,\{[(z_1-1)\tilde{L}_{TP} + E(\tilde{L}_{TP})] - [(z_1-1)\tilde{L}_{DRP} + E(\tilde{L}_{DRP})]\} + \sum_{3}^{n-1} r_i z_i L_{ISPi} + \sum_{n}^{N} z_i L_{DSPi}. \qquad /5/$$

Here L_T and L_{DRP} are river total - and dissolved reactive phosphorus loads, resp., L_{ISPi} indirect sewage load[1], L_{DSPi} direct sewage load and"a"expresses the available fraction of particulate P load. r_i are retention coefficients $/0 \leq r_i \leq 1/$ defining the portion of P which reaches the lake from the indirect sewage discharges at given points of the tributary."E" refers to the expectation of a certain probabalistic variable.

1/ Phosphorus of sewage is assumed to be completely available for algal growth.

In Eq/5/ the first two terms expresses the reduction effect of the second reservoir stage $/z_2/$ on the expectation and fluctuation of $\tilde{L}_{DRP}$. The third term reflects the influence of sedimentation in the first reservoir element, while last two terms are related to indirect and direct sewage control.

Apparent from the equation is, that tertiary treatment affects only the expectation of the load, while reservoirs have impact on both expectation and fluctuations. Of the $E/\tilde{L}_{DRP}/$ load $\sum_3^n r_i z_i L_{ISPi}$ is of sewage origin. Thus, the nutrients carried by tributaries can be controlled by both tertiary treatment and reservoir projects. Value of z_2[1/] is constrained by the physical condition

$$z_2 E\ /\tilde{L}_{DRP}/ + \sum_3^n r_i z_i L_{ISPi} \leq E\ /\tilde{L}_{DRP}/ \qquad /6/$$

/no more nutrients can be removed than those originally loading the lake/. The same limit is not valid for z_2^* : $0 \leq z_2 \leq z_2^* \leq 1$.

It is stressed that Eq/5/ operates on an annual basis. With Eqs/4/ and /5/ we established a direct relationship between water quality and control variables.

4.4. Objective function

Let us define the water quality goals by Y_{gj} $/1 \leq j \leq 4$ here/. We would like to reach the goals under stochastic changes and uncertainties as closest as possible which expectation can

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1/ Note that z_2 is related to the original load $E/\tilde{L}_{DRP}/$ rather, than to the actual influent load which depends on z_i $/3 \leq i \leq n/$. The latter formulation would cause non-linearity between z_2 and z_i $/3 \leq i \leq n/$ in Eq./5/.

be expressed by the following deterministic objective function

$$\min \sum_{j=1}^{4} W_j \; E \, /\tilde{Y}_j - Y_{gj}/^{w}. \qquad /7/$$

If w=2, the weighted sum of variances is minimized for four basins. If w > 2, deviations from goals are penalized at a larger extent. Through W_j some of the basins can be preferred over others /more intensive tourism or need for urgent control etc/. In general, the inclusion of parameters w, W_j and partially also Y_{gj} allows the analyst to handle the subjective judgement of users, knowledge of decision makers hard to quantify and various aspects of the policy making procedure.

From the formulation of Eq./7/ and the "soft" character of the management problem it follows that one has quite large flexibility /and arbitrariness/ to define the objective function in order to derive "feasible" solutions. In the context of the present work Eq./7/ was replaced by an analogious objective $/\Delta\tilde{Y} = Y_o - \tilde{Y}/$

$$\max \sum_{j=1}^{4} W_j \; E \, /\Delta\tilde{Y}_j/ - w\,\sigma/\Delta\tilde{Y}_j/ \qquad /8/$$

i.e. the combination of expectation and standard deviation of water quality improvement is maximized. Eq./8/ can be set forth as a function of control variables with Eqs./4/ and /5/ after deriving analytically /through some approximations/ expectations and standard deviations by using the actual gamma, log-normal, normal etc. distributions outlined in previous sections. The advantage of Eq./8/ is that it maintains the major features of Eq./7/ but results in a linear model.

Provided that $a_{11}...a_{44} = 1$, $a_{21}...a_{34} = 0$ in Eq./2/ the objective function is formulated in terms of the volume related loads.

4.5. Constraints

In order to select among the alternatives of different investment cost, IC and operation, maintenance and repair cost, OC, the term total annual cost, TAC is used

$$TAC = \sum_i OC_i + \sum_i q_i IC_i \qquad /9/$$

where q_i is "capital recovery factor" depending on the discount rate and length of the project's life /see e.g. Loucks et al 1981/. This factor can be different for "small" projects /chemical P precipitation/ and "large" projects /reservoirs of considerable sizes/, resp.

In most cases TAC is limited by economic conditions. Additionally, it is possible to limit the total investment-, or operation cost. Certainly, a multiobjective approach can be easily employed, too, by incorporating TAC into the objective function Eq./8/ /maximal water quality reduction at "minimal" total annual costs/.

Other constraints /upper and lower bounds, furthermore logical and physical conditions/ were discussed earlier.

4.6. Solution technique and evaluation of results

The linearised model arrived at after all these steps was solved by using a conventional linear programming routine. No mixed integer programming was applied; the small number of integer variables were fixed in the input on the basis of experiences gained previously with the continuous model.

The model results in among others the expectations and standard deviations of basins'water quality. Distributions of water quality are then evaluated in the frame of a Monte Carlo simulation /1000 runs were done in each case/. At this step of the analysis the influence of smaller tributaries /which were disregarded in the optimization/ is taken into account, too.

Finally, it is noted that the eutrophication management model derived which is based on thorough scientific analyses is now simple and fast enough on computer for interactive usage /excluding Monte Carlo simulations/ being an effective tool in the frame of the policy making procedure. Checking the results with dynamic lake eutrophication model as shown on Fig.1. can be obviously performed outside the interactive use, only.

5. RESULTS FOR LAKE BALATON: SHORT TERM CONTROL

As the basic situation, $r_i = 1$ /no retention in rivers/, $w_j = w = 1$ /equal weighting/, $q_i = q = 0.1$ /equal discounting/ and no effluent standard prescription was considered, and optimization was performed under different TAC conditions /TAC = $2.5 - 25 \cdot 10^7$ forint, Ft/. Results for the Keszthely basin /expectation, standard deviation and extremes for water quality indicator $Y = /\text{Chl-a}/_{max}/$ gained from Monte Carlo procedure are given in Fig.7. as a function of TAC[1].

Depending on the total annual cost[2], four domains of solutions can be distinguished:

/i/ In the range of TAC = $0-5\cdot 10^{7}$ Ft sewage treatment can be intensified and tertiary treatment introduced. Expectation of concentration will decrease considerably, but not the fluctuations. Under very small costs / $\sim 3\cdot 10^{7}$ Ft investment cost/ only the sewage of Zalaegerszeg /Fig.6/ should be treated even if the retention on the Zala River is larger than 50%. Under increasing budget the investments are accomplished from west to east.

/ii/ If TAC moves between $5\cdot 10^{7}$ and $10\cdot 10^{7}$ Ft, the effectiveness of sewage treatment can not be increased further but reservoir systems are still too expensive.

/iii/ At about TAC=$15\cdot 10^{7}$ Ft the solution is a combination of tertiary treatment and reservoirs. Fluctuations in water quality are strongly reduced by the latter control alternatives.

1/ Note that strictly meaning $/Chl\text{-}a/_{max}$ is not a continuous function of TAC due to the presence of integer variables. However, a precise interpretation would cause only slight changes in Fig.7.

2/ Running cost ranges between $0.5\cdot 10^{7}$ and 10^{7} forint /40 Ft is about 1 US dollar/, thus beside q=0.1 total investment cost is roughly ten times larger than TAC.

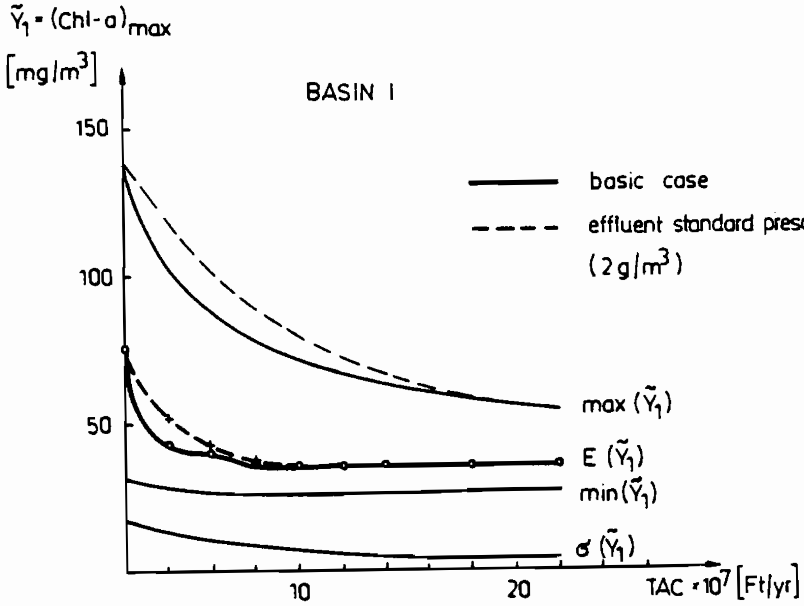

Fig. 7. Water quality as a function of the total annual cost /w_j = 1, w = 1, r_i = 1, q = 0.1/

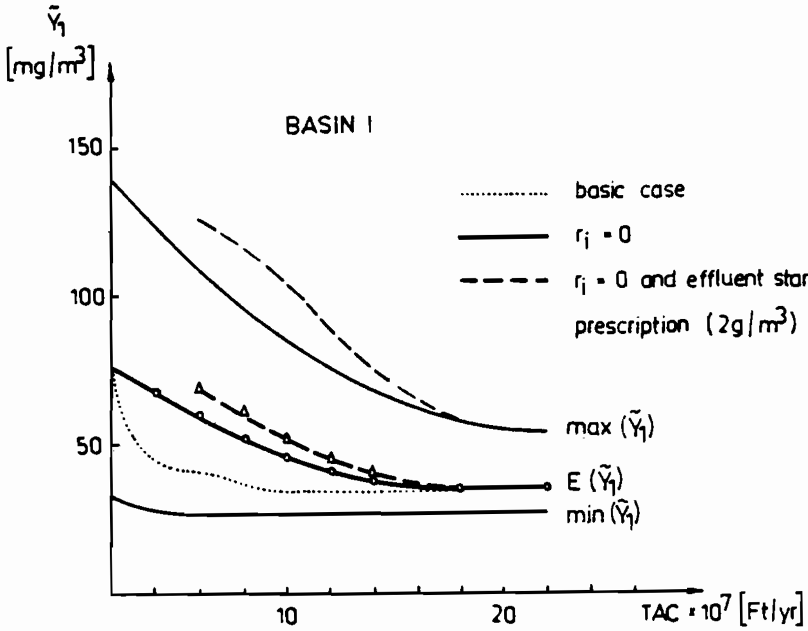

Fig. 8. Sensitivity of the solution on P retention coefficients and effluent standard presciption

/iv/ Finally if TAC $\geq 20 \cdot 10^7$, tertiary treatment is dropped on regions where reservoirs form an alternative.

In Fig.8. the influence of river retention coefficient and effluent standard is analyzed. The conclusion is quite obvious: under larger phosphorus retention and given effluent standard criterion /$2 g/m^3$/ the improvement in water quality is less remarkable as compared to the basic case /Fig.7/. The worst situation is if all the phosphorus is removed along the river and still treatment has to be performed. In this case a portion of the budget should be allocated for investments having no influence on the lake's load. This clearly shows that a water quality criterion / $2 g/m^3$ in this case/ independent of the major features of the system can result in a strategy far from the optimal.

Fig 7. and 8. illustrates primarily the sensitivity of the solution on basic parameters. In reality the retention coefficients are about 0.7 /see Jolánkai and Dávid, this proceedings/ /$\sim$ 0.5 for Keszthely the sewage of which is diverted to a marshland/. Under such conditions /Fig.9./ tertiary treatment affects the expectation of water quality at a slightly smaller extent than in the basic case /Fig.7/. Through phosphorus precipitation $\check{Y}_1 = (Chl\text{-}a)max$ is reduced to 50-55 mg/m^3 but still extremes larger than 100 mg/m^3 can occur /hypertrophic domain/. When accomplishing reservoirs, too, the corresponding values are 30-35 mg/m^3 and 60 mg/m^3, resp. /eutrophic stage/.

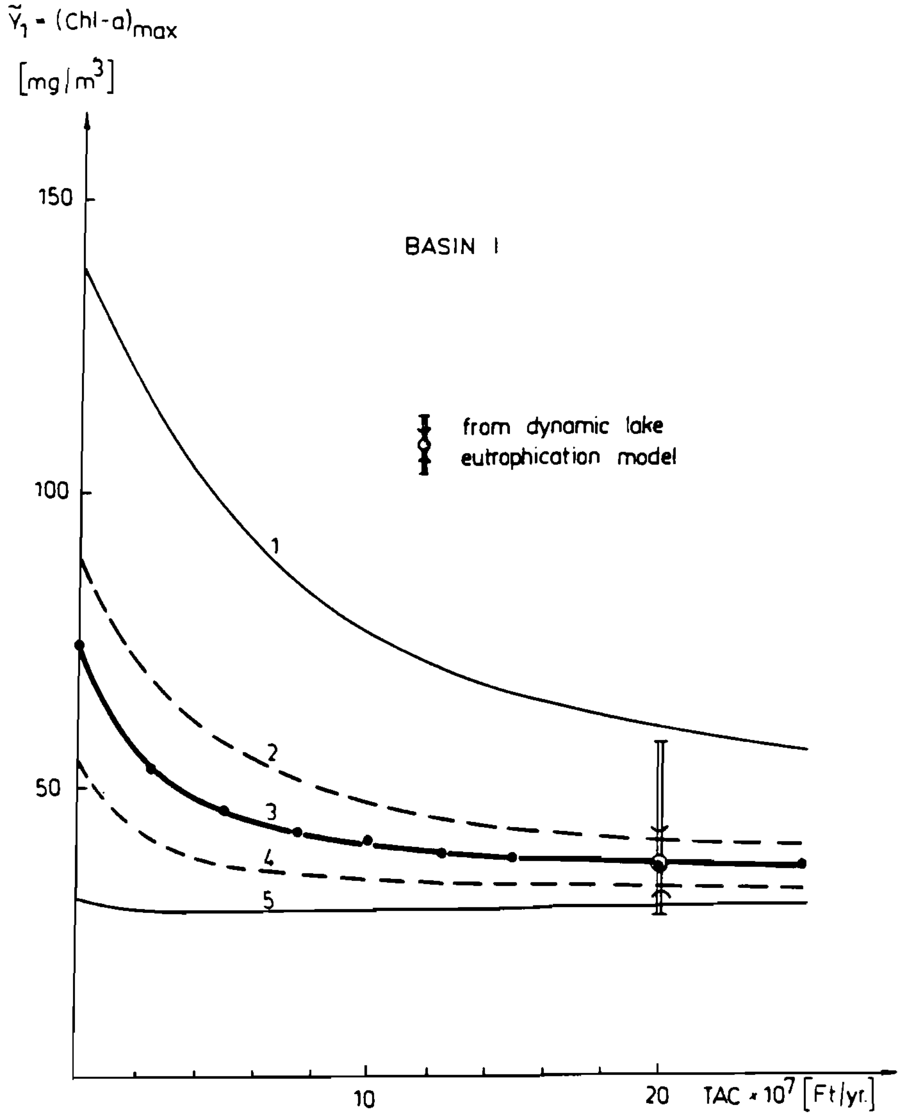

Fig. 9. Model solution with actual P retention coefficients

3 - expectation, 2 and 4 - ± standard deviation,

1 and 5 extremes /all from 1000 Monte Carlo simulations/

In Fig. 9. the results of the detailed simulation model for an optimal solution /TAC = $20\cdot10^7$ Ft/ is also given. The satisfactory agreement indicates that the aggregated lake eutrophication model is quite proper for our present objective.

Fig.10. compares the probability density functions of two considerably different solutions /TAC = $2.5\cdot10^7$ and $25\cdot10^7$ Ft, resp./ for four basins derived from Monte Carlo simulations. Also from this figure we can conclude that tertiary treatment is more effective than reservoirs /where they form alternatives/ for the mean concentration, but fluctuation can be controlled only by reservoirs /though investment costs are larger/. Consequently short term strategy of eutrophication management should focus on tertiary treatment /where diversion is not completed in the nearest future/ from west to east, with a faster realization of reservoir projects already planned.

Concerning the major parameters of the model, the following can be stated.

/i/ From Fig.7. and what we already said about the use of water quality standards it follows, that for lakes consisting of basins with different water quality the minimization of the load does not result in an optimal control strategy. As was also stated by Hughes /1982/ the volume related loads are to be minimized at least.

/ii/ Retention coefficients are extremely important from the viewpoint of decision making. If the P retention is

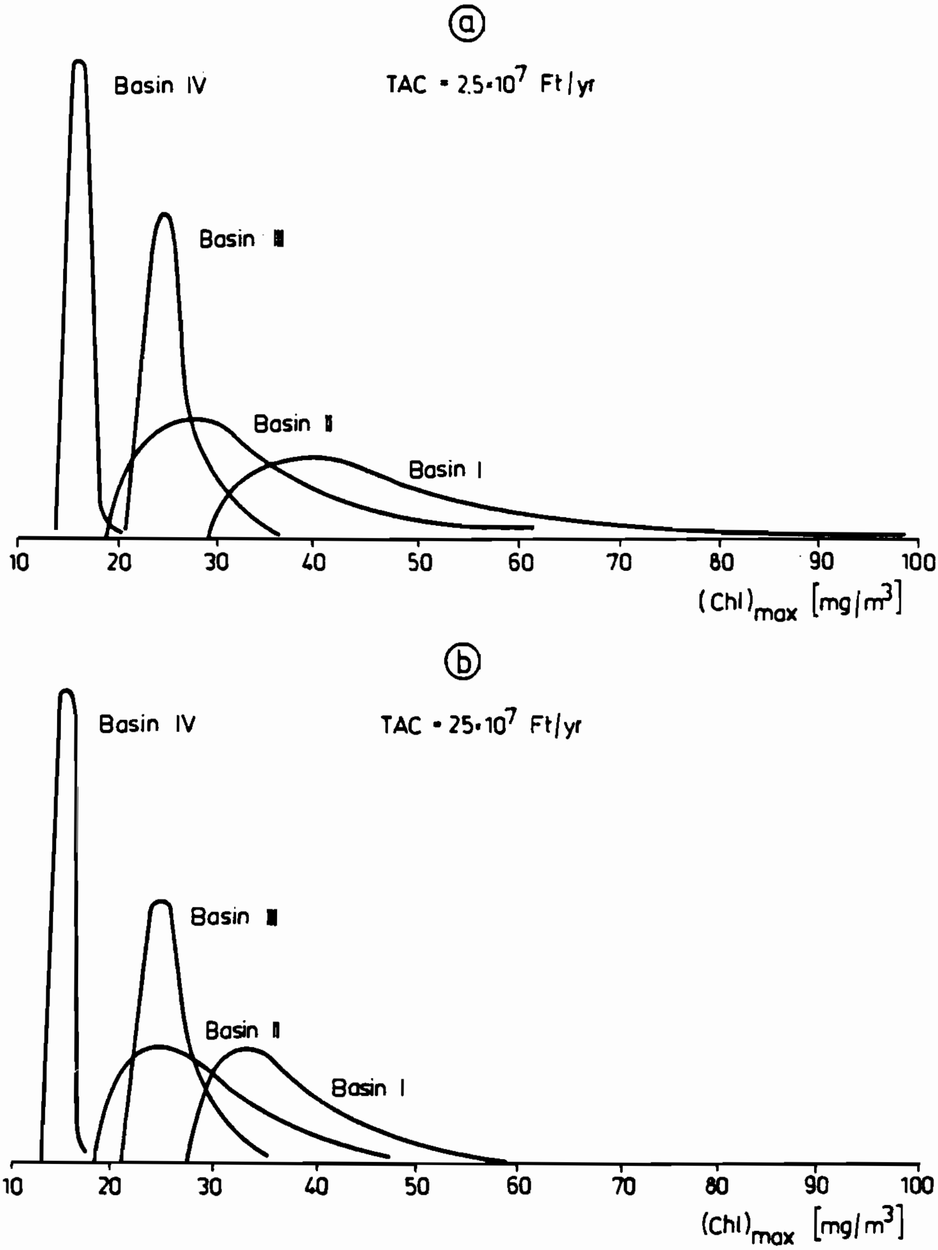

Fig. 10. Probability density functions for two different situations
/derived from 1000 Monte Carlo simulations/

100 % in rivers, reservoirs are the only effective control tools.

/iii/ the P removal mechanism and efficiency of reservoirs are not quite satisfactority explored /Uhlmann and Benndorf 1980, Burton et al 1979, Fetter et al 1978, Tóth 1979/. It is stressed however that uncertainties in cost estimates play quite the same role on decision making than in removal rates.[1] Both of them shift the particular TAC value where reservoirs appear in the solution.

/iv/ Capital recovery factor/or discount rate/ has a similarly important role. In general, the conditions of borrowing money is much more advantageous for large investments /such as the Kis-Balaton system of a surface area of about 70 km^2/ than for small ones. For these projects $q = 0.05$ is more realistic than 0.1 which means that reservoir projects start to be feasible "sooner", than shown on Fig. 7.

/v/ If w=0 /deterministic treatment/ reservoir projects enter the solution under larger budget conditions as compared to w=1.

/vi/ As was shown by Baker and Harleman /this proceedings/ the slope of the load response line /$a_{11} \ldots a_{44}$ in Eq./2// is a subject to uncertainties, too. In this particular case the solution of the management model has a limited sensitivity on this factor, only.

/vii/ In extreme cases the above slope can be very small

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[1] This fact reveals that cost-benefit analyses are as important as studying different technologies or various biological, chemical etc. subprocesses in the lake and watershed.

for some of the basins /internal load $\gg$ external load/ calling for no management actions for the sub-regions associated which would be irrealistic. In such situ.tions matrix $\underline{\underline{A}}$ should be replaced by $\underline{\underline{A}}^{*}$ using the coefficients $k_1 \ldots k_4$ as formal parameters in the frame of a sensitivity analysis. Another possibility is just to use the volume related load for the purpose of optimization. This is equivalent to dropping the lake eutrophication model completely on the level of management; a fact which stresses again the need on sediment research.

6. OTHER MANAGEMENT MODELS DEVELOPED

Major features of models developed for Lake Balaton are summarized in Table 1. The starting point of Model 1 is the cause-effect relation of eutrophication: a direct relationship is established on the basis of historical data between water quality and the factors influencing nutrient loads /e.g. fertilizer use, tourism, sewage discharges/. The approach distinguishes four lake-basins and seven connected subwatersheds. Although the model does not incorporate explicitely economic considerations, it is a useful, simple tool for long-term planning.

In harmony with the findings of the previous section, results of Model 1 clearly indicate that protection measures taken in the Zala watershed are about 6 times more effective than those in the most eastern subwatersheds. A remarkable

No.	Model	Technique	Application	Comments
1.	Watershed development approach Dávid and Telegdi /1982/	Regression and trend analysis	Entire lake-watershed system	No budgetary considerations
2.	Multiobjective analysis Bogárdi et al /1983/	Stochastic nutrient load model and multiobjective programing	Tetves subwatershed / $\sim$ 300 km^2/	Lake's water quality is not included. Emphasis on stochastic features and methodology.
3.	Sewer system planning Kovács et al /1983/ /this proceedings/	/0,1/ integer programing	Recreational area	Engineering type planning of sewer network with waste water treatment plants. Constraints for effluent concentrations, but lake's water quality is not considered.
4.	Optimal spatial configuration of tertiary sewer-, and reservoir projects Hughes /1982/	/0,1/ integer programing	Entire lake-watershed system	Deterministic load response model is incorporated
5.	Short term control strategy Somlyódy /this paper/	Linear programing	Entire lake-watershed system	Stochastic load response model is used

Table 1. Eutrophication management models developed for Lake Balaton

feature of the approach is that relatively few, mainly statistical data are required being generally available already at the beginning of such a study.

The economic objective of Model 2 is to minimize the sum of costs and losses for the various control measures /alternatives are: point source-, fertilizer-, erosion-, land use- and runoff control, furthermore sediment trapping/,and environmental objective is to minimize the amount of biologically available phosphorus load.

Results gained for a small southern agricultural subwatershed of Lake Balaton stress the importance of stochastic nature of phosphorus load. For the particular situation considered the combined control measures consisting of storage, sediment trapping and tertiary treatment seemed to be the best compromise: similar conclusion as was drawn in Section 5.

Model 3 considers the development and planning of sewer system in the recreational area. Preliminary results showed interesting results in relation to the question of local versus regional sewerage systems /Kovács et al in this proceedings/. According to the model 2-3 regional treatment plants are preferred on the southern shore but about 10 on the northern due to the high cost of the network /hilly region/ and relatively high percentage of existing sewerage.

Model 4 has approximately the same, though simpler structure than Model 5 discussed more in details in this paper. It incorporates the deterministic load response model given by Eqs./1/ and /2/, and uses /0,1/ decision variables.

Conclusions are similar to that given in Section 5:

- control measures should start at the western end of the lake /as stated also by Model 1/;

- sewage projects /tertiary treatment/ are more cost effective than reservoirs where they form alternative options /an exception is perhaps the largest reservoir system, Kis-Balaton/;

- minimizing the load is quite a different objective than maximizing the improvement in lake water quality /due to nutrient load reduction/. This is generally true for lakes having segments of highly different water quality.

Model results outlined here briefly confirmed the findings discussed in details in Section 5.

7. CONCLUSIONS

/i/ Based on sound scientific analysis it is possible to establish an eutrophication management optimization model as a powerful tool for decision making which preserves the influence of the most important in-lake-, and watershed subprocesses in an aggregated manner. The advantage of such an approach is that lake water quality desired to achieve through control measures can be directly involved in the model formulation.

/ii/ When describing the various biological, hydro-physical etc. subprocesses several shortcomings in under-standing appear. In the course of developing the water quality management model many details of these processes are ruled out. For instance algal dynamics is of less importance and the annual peak chlorophyll-a concentration, $/Chl\text{-}a/_{max}$ can be used as an indicator of eutrophication. Thus the usefulness and effectiveness of management model as described under item /i/ depends on at what extent the scientific gaps are still preserved.

For Lake Balaton a stochastic, linear load response model was derived for $/Chl\text{-}a/_{max}$ by using a dynamic lake ecological model. The most important parameters of such an aggregated model are representing sediment release /internal load/ and interbasin mass exchange /control measures taken on subwatersheds of a certain basin can influence the water quality of other basins/.Probably similar aggregated lake model can also be derived for many other shallow lakes having regions of considerably different water quality.

Lack of knowledge on sediment and interbasin transport can hamper the use of such aggregated lake model and the formulation of management objectives in terms of lake's water quality. Minimization of a combination of volume related loads /but not the loads themselves/ of individual basins is the best what can be done under such circumstances.

/iii/ For Lake Balaton the objective was to establish the "optimal" short term control strategy. For this purpose the model describing the "immediate" response of the lake for load reductions under various stochastic influences and uncertainties were used. Handling of interbasin transport did not cause difficulties as the basins are unidirectionally coupled and practically only the neighbouring segments influence each other. The sensitivity of the solution of management model was not significant on the present level of internal load. Nothing can be said, however on the equilibrium concentration of the lake /long term response/ and the time required to reach this state. Much more knowledge would be necessary on the behaviour of sediment to answer these questions.

Stochastic properties of nutrient loads and meteorological factors, and uncertainties /e.g. related to infrequent sampling on tributaries/ are of major importance on the level of decision making. The situation should be the same for other shallow lakes.

/iv/ Various management models developed clearly indicated that control measures should be realized from west to east. The short term control strategy should focus on the introduction of tertiary treatment on the existing plants /under unchanged sewer network/. Tertiary treatment is effective in reducing the expectation of the water quality, but fluctuations are about the same as without control. The reduction of random changes in water quality requires the establishment of pre-reservoirs equalizing the variations in loadings deriving mainly from non-point sources. Investment costs are considerably larger in this case, however.

/v/ Beside parameters representing in-lake processes several other factors of equal importance are involved in the management model. Of these river retention coefficients, effluent standards, cost functions and their uncertainties, and capital recovery factors which can differ for sewage and reservoir projects, resp., should be mentioned. Prescription of overall water quality standards for lakes of non-uniform water quality in space often results in control strategy far from optimal.

The subjective factors of the policy making procedure are also significant /e.g. how to rank the reduction in the mean and variance of concentration, resp.; whether various regions of the lake are or are not equally important from the viewpoint of water use/.

/vi/ Structural changes of water ecosystems can not be satisfactority predicted on the present level of knowledge. On the other hand the gradually increasing pollution of lakes causes more and more water quality problems to be solved within a relatively short period of time. As in the solution "microscopic" /mostly "scientific"/ and "macroscopic" /practical, economic etc./ issues appear approximately with the same importance, the most feasible approach is to start studying the problem at the same time from both, microscopic and macroscopic angles. Results should be then gradually integrated and offered updatedly for application. Otherwise the danger occur that for the completion of the study the system already changed structurally and conclusions are of no practical use.

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I would like to thank my colleagues at IIASA, primarily R. Wets, D.P. Loucks and T. Hughes, with whom I had fruitful discussions and who supplied valuable comments on the particular management model discussed in more detail in this paper.

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EPILOGUE

The problem of Lake Balaton raised numerous scientific and managerial questions. The present work was perhaps broader than other lake water quality studies as it covered on an integrated manner more disciplines ranging from biochemistry to economics all related somehow to man-made eutrophication; a problem of the lake and its surrounding region. This "horizontal broadness" of the study was associated with "deepness" whenever required on individual disciplines due to the participation of Hungarian research institutes having historical background on various fields and access to in situ and laboratory experiments.

Retrospectively, many research gaps were clarified. For example, a far better understanding is available up to now in general on the relative importance of hydrophysical and biochemical processes in eutrophication for lakes of spatially strongly variable water quality than before. The knowledge on sediment and wind induced sediment-water interaction was improved, too. A reliable estimate was gained for external nutrient loadings. The importance of various uncertainties was recognized and included in the analysis. These uncertainties are partially related to the scarcity of observations demonstrating the crucial role of properly designed monitoring system. A considerable progress was achieved on understanding biochemical processes by the synthesis of existing data and further experimental and modeling work. Various management models clearly indicated which are the most urgent measures to be taken for protecting the lake's water quality. Finally, several methodological questions were elucidated.

The Case Study had other achievements, too. Several communication problems were resolved. Scientistis of different backgrounds who previously used to work more or less independently on various aspects of the same multidisciplinary problem established contacts and started to cooperate: a very positive outcome. Also, a fruitful interaction was realized among scientists of different nations who participated in the research. The problem of Lake Balaton raised the interest of several researchers: the lake is still a subject of study in foreign countries too, such as in the USSR, USA, UK, the Netherlands and Finland.

The communication gaps between scientistis and decision makers were also narrowed. The results of the study were continuously discussed with representatives of Hungarian institutions responsible for the lake. In May,1982 a Colloquium was held at IIASA where the practical conclusions of the research were overviewed and discussed with some top-level Hungarian decision makers. As a follow-up a meeting between scientists and decision makers and their supporting staff was organized in the closing workshop. Simultaneously, the results of the Case Study were utilized in the ongoing policy making procedure in Hungary leading eventually to a government decision on water quality control and regional development this January (see Preface) and subsequently to the modification of the existing management action plan.

From the foregoing one may conclude that the analysts clarified all the dark spots. This is not the case. In contrary, several questions remained - at least partially - non-resolved and plenty of further research gaps were discovered. Most of our positive statements mentioned before can be complemented by corresponding shortcomings. For instance, the understanding of turbulent motion in shallow lakes is still far from complete (although it seems to be satisfactory as far as the effect of this motion on changes of water quality is concerned). The ecological consequences of wind induced resuspension and other processes associated are not so well identified. Sediment forms one of the major research gaps in understanding and controlling eutrophication. The stochastic nature of nutrient loads and the identification of individual pollutant sources should be explored more accurately. Analytical techniques in biochemistry have to be refined (sediment research, orthophosphate determination, measurement of reaction rates, etc). Better understanding is required on the structural changes of ecosystems.More knowledge is needed on technologies for nutrient removal including the related costs and benefits, and the list is certainly not complete.

In a summary, the study resulted in a number of useful conclusions, both Lake Balaton oriented and general in character. Methodologically the research showed how effective a systems analytical approach can be for studying a multidisciplinary, complex problem of a large system where uncertainties play an important role and both scientific and practical issues have to be handled. Future research needs were also identified.

For the particular case of Lake Balaton, perhaps three major classes of future research areas should be mentioned. The first class is formed by "microscopic" issues such as the problem of blue-green algae prolificated in the lake in 1982 and the sediment. The second group involves themes more "macroscopic" in character, e.g. the regional development and the analysis of socio-economic changes in the Balaton area. Finally, the third category of research should focus on the monitoring of the effectiveness of water quality protection measures which were decided upon this year.

Part II

Selected Papers and Summary of the State-of-the-Art Discussion

BEM: A COMPLEX MODEL FOR SIMULATING THE LAKE BALATON ECOSYSTEM

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INTRODUCTION

The modelling of such a complex ecosystem like Lake Balaton with the problem of eutrophication dictates two goals: first to improve the model for better understanding the system behaviour and for better simulating measured data, second to make the model suitable for management purposes, to raise basis for decisions. These contradictory goals result that the present model is the last in the series of models but not the final one, as the model develops together with the progress of ecological knowledge, while the second goal needs strong interactive cooperation with decision makers and modellers. The philosophy of modelling hasn't changed during the time: the structure of the model has to reflect the structure of the ecosystem, the main processes in the lake have to be involved into the model, the parameters in the mathematical description have to carry biological meaning and the model has to be able for management purposes, too. The lake had been divided into four basins based on hydrodynamical experiments and considerations (Baranyi, 1979). These basins are modelled separately and the models of the basins are concatenated by hydrological throughflow. At present all the basins are described by the same model and parameter set.

MODEL DESCRIPTION

The model describes the main mass transport processes in the open water of the basins. As the distribution of matter in open water can be regarded uniform, the open water is described by one point. The state variables are concentrations and biomass in one liter of water. The matters in the sediment taking part in matter transport are referred to one liter of water column over the sediment for simplifying calculation. The concentrations and biomass are expressed in phosphorus or in nitrogen or in organic matter. The organic matter produced by primary production contains phosphorus and nitrogen in constant ratio according to the model and the ratios are the same in the release of phosphorus and nitrogen during the decomposition of organic matter. So the mass conservation is valid for phosphorus and nitrogen in the model.

The forcing functions of the model are global radiation water temperature and nitrogen and phosphorus load coming from the watershed of the lake. The model contains ten state variables for one basins: winter-spring, summer and autumn phytoplankton, blue-green algæ , bacterioplankton, dead organic matter, dissolved inorganic nitrogen, organic matter and phosphorus pool in the sediment (see Fig.1.).

According to the model nutrients circulate in two main cycles within the basin. In the first one the inorganic phosphorus and nitrogen are built into organic matter by algæ during primary production and the mortalized algæ become dead organic matter which is decomposed by bacterioplankton revealing the inorganic nutrients. In the second cycle some dead organic matter, algæ and bacteria settle down, and after decomposition in the

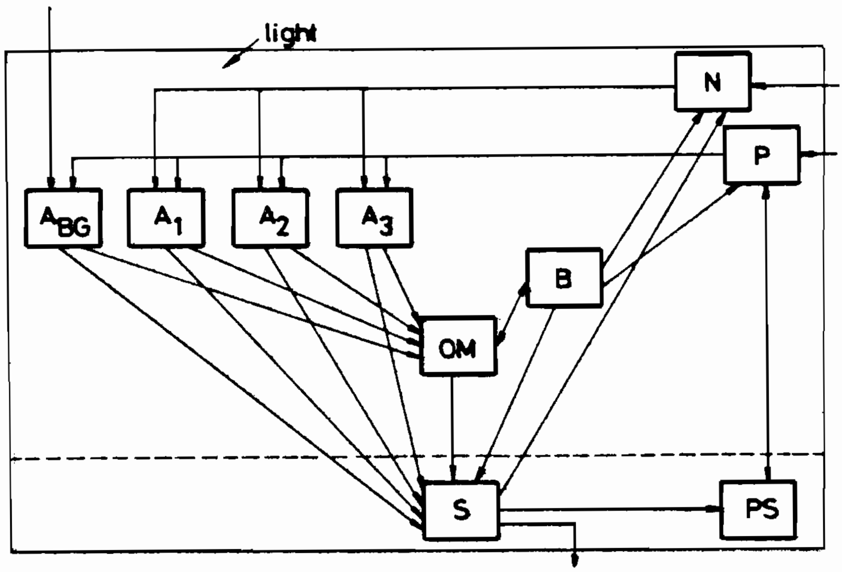

Fig.1. Diagram of model compartments

sediment phosphorus and nitrogen are released. The nitrogen gets directly back to the water, the phosphorus gets into the exchangeable phosphorus pool of the sediment and from this pool it comes to the water. Nutrients can leave the system not only by outflow but by stabilization in the sediment and nitrogen can disappear by denitrification too. More detailed description of the ecological system can be read in the article of Herodek, Kutas and Csáki (1983).

For all the four phytoplankton compartments the formulation of primary production is the following:

$$PROD = PMAX \cdot TEMP \cdot U \cdot A$$

where PMAX is the maximal production rate, A is the algal biomass, TEMP is the temperature limitation in the following form

$$\mathrm{TEMP} = \begin{cases} \dfrac{\mathrm{TCRIT} - \mathrm{T}}{\mathrm{TCRIT} - \mathrm{TOPT}} \cdot \exp\left(1 - \dfrac{\mathrm{TCRIT-T}}{\mathrm{TCRIT-TOPT}}\right) & \mathrm{T} \leqslant \mathrm{TCRIT} \\ 0 & \mathrm{T} > \mathrm{TCRIT} \end{cases}$$

where TCRIT and TOPT are the critically high and optimal temperature respectively, T is the water temperature. U is the joint limitation of nutrients and light. For the first three phytoplankton compartments it has been formulated in the following way

$$U = (U_1 \cdot U_L)^{U_\circ} \cdot U_\circ^{(1-U_\circ)}$$

where

$$U_1 = \min(U_P, U_N)$$

$$U_P = \frac{P}{HP+P}$$

$$U_N = \frac{N}{HN+N}$$

where P and N are inorganic phosphorus and nitrogen concentration in the water, HP and HN are half saturation constants. U_L is the light limitation in the following form

$$U_L = \frac{1}{Z\ EXT} \cdot \left[\exp\left(1 - \frac{LZ}{LOPT}\right) - \exp\left(1 - \frac{L_\circ}{LOPT}\right)\right]$$

$$LZ = L_\circ \cdot \exp(-Z \cdot EXT)$$

where Z is the depth of the waterbody, EXT is the extinction coefficient, LOPT is the optimal light intensity and $L_\circ$ is the global radiation on the water surface.

$$U_\circ = \min(U_1, U_L)$$

The primary production of blue-greens has been formulated in a similar way only the U term is different. As blue-green algae can fix nitrogen of the air nitrogen doesn't limit the primary production, blue-greens can float in the water in the optimally lightened layer thus neither the light limit the production. In this case the U term reduces to phosphorus limitation ($U=U_P$).

The natural mortality of all living compartments is the exponential function of the water temperature

$$MORT=MCRIT \cdot \exp\left[MR \cdot (TCRIT-T)\right] \cdot B$$

where MCRIT is the mortality rate at critically high temperature, MR is the temperature coefficient of mortality and B is the biomass of proper compartment. The dead organic matter uptake of bacteria has the following formula

$$UPT=CMAX \cdot \frac{OM}{OM+B} \cdot TEMP \cdot B$$

where CMAX is the maximal uptake rate, OM is the concentration of dead organic matter in the water, B is the biomass of bacterioplankton and TEMP is the temperature limitation in the same form as at the primary production. The decomposition process is based on TEMP term as well

$$DEC=DECR \cdot TEMP \cdot B$$

where DECR is the decomposition rate and B is the biomass of bacteria. A part of it appears in the equation of phosphorus and another part does in that of the nitrogen as excreted phosphorus and nitrogen

$$EXP = PR \cdot DEC$$

$$EXN = NR \cdot DEC$$

where PR and NR are the phosphorus and nitrogen ratios of organic matter and they are supposed to be constant in every form of organic matter. These values are used in the uptake of inorganic nutrients

$$PUPT=PR \cdot (PROD_{A1}+PROD_{A2}+PROD_{A3}+PROD_{BG})$$

$$NUPT=NR \cdot (PROD_{A1}+PROD_{A2}+PROD_{A3}) \quad .$$

As the blue-greens can fix the nitrogen of the air they don't consume the nitrogen of the water, so this is a special nitrogen input for the system.

The sedimentation of all particulate materials is proportional to their biomass or concentration

$$SED = SR \cdot B$$

The decomposition of organic matter in the sediment is the exponential function of temperature

$$MIN=MINR \cdot \exp(STEMP \cdot T) \cdot S$$

where MINR is the mineralization rate, STEMP is the temperature coefficient of the mineralization and S is the concentration of organic matter in the sediment. Laboratorial experiments (Istvânovics, Herodek, Entz, 1982) revealed that the nitrogen release of sediemnt showed high temperature dependence while the same dependence wasn't observed in the case of the phosphorus. The experiments with phosphorus referred to adsorption-desorption process. This was modelled in the following way: a part of decomposed organic matter gets into the phosphorus pool of the sediment

$$SUPL = PR \cdot MIN \cdot (1-SFR)$$

where SFR is the stabilized fraction. The phosphorus exchange between the sediment and the waterbody is the following

$$EXC = EC \cdot (PSC \cdot PS - P)$$

where EC is the phosphorus exchange coefficient, PSC is the sediment phosphorus coefficient and PS is the phosphorus concentration in the sediment. This term appears with negative sign in the equation of sediment phosphorus pool, while with positive sign in that of phosphorus of the water. The nitrogen gets to the water directly

$$RELN = NR \cdot MIN \cdot (1-SFR)$$

but a part of it leaves the system during the process of denitrification

$$DENIT=NR \cdot MIN \cdot (1-SFR) \cdot \left[1-\exp(SRN \cdot MIN)\right]$$

where SRN is the tempereature coefficient of denitrification. Contrary to assumptions having been supposed the nitrogen plays a very important role in the ecosystem, not only in temporary processes but in the eutrophication process, too.

When the lake is covered with ice the conditions under it change as there is no wind induced stirring up. In the model this was simulated by greater settling rate in this period. As a consequence a thick algal carpet develops on the bottom. After ice breaking large amount of accumulated organic matter gets into the water. This can be deduced from winter observations (Herodek and Olâh, 1973) and the very high concentration of nitrate in early pring. This stirring up process was simulated by getting a part of organic matter of the sediment into the water as inorganic phosphorus and nitrogen. The whole thing takes place during one month beginning one week after ice break.

A detailed description of process formulations and the equations of BEM model can be read in the Working Paper of Kutas and Herodek (1982). The equations of the BEM model can be seen on Table 1.

Table 1. Equations of the BEM model.

$$\frac{dA}{dt} = PROD_A - MORT_A - SED_A + INFLOW_A - OUTFLOW_A$$

$$\frac{dB}{dt} = UPT - DEC - MORT_B - SED_B + INFLOW_B - OUTFLOW_B$$

$$\frac{dOM}{dt} = \sum_{i=A_1 A_2 A_3 BG, B} MORT_i - UPT - SED_{OM} + INFLOW_{OM} - OUTFLOW_{OM}$$

$$\frac{dS}{dt} = \sum_{i=A_1 A_2 A_3 BG, B} SED_i - MIN$$

$$\frac{dPS}{dt} = SUPL - EXC$$

$$\frac{dP}{dt} = LOAD_P - PUPT + EXP + EXC + INFLOW_P - OUTFLOW_P$$

$$\frac{dN}{dt} = LOAD_N - NUPT + EXN + RELN + INFLOW_N - OUTFLOW_N$$

SIMULATION RESULTS

The results of the model have been compared to measured chlorophyll-a, the only characteristic of water quality which had been measured simultaneously and quite frequently in all the basins (see Fig. 2-5.). To get chlorophyll-a the phytoplankton biomass, which is the state variable of the model was multiplied by a constant (17.837) estimated from paralel biomass and chlorophyll-a measurements (Herodek, Vörös and Tôth, 1982), the biomass has been expressed in mg dry weight/l while chlorophyll-a in µg/l.

According to measurements the chlorophyll-a concentration had a spring and a summer peak. In Keszthely and Szigliget basins the spring peak was the highest in the first year and the summer one was the highest in the second year caused by a water bloom. In the other two basins the spring peak was the highest. The model could simulate these peaks in Szemes and Siôfok basins but the spring peak slipped to an earlier date than it had been measured. This is due to the problem of ice the uncleared circumstances of the effects of ice breaking process on the ecosystem cause that the spring algal bionass peaks in simulation runs appear earlier than in reality.

In the first and second basins the spring peaks have been simulated properly with the same problems we had in the other basins. The summer peak in the first year have been over-estimated and in the second year under-estimated, the simulated curves have approached the water bloom chlorophyll-a curves, but they haven't reached them in the second year. The eruption of chlorophyll-a curves at the end of the simulated period are due to a false ice breaking process.

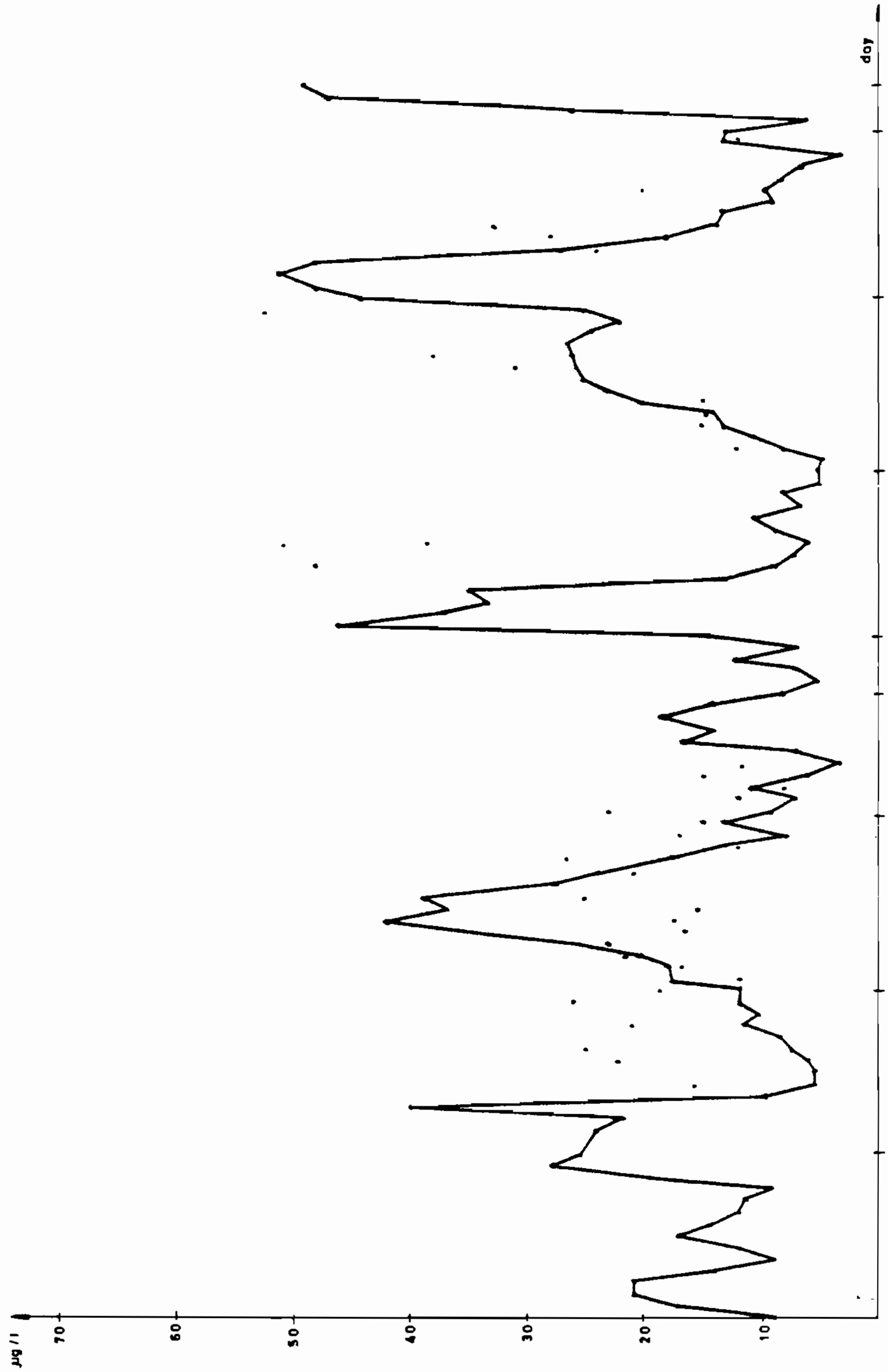

Fig. 2. Measured and simulated chlorophyll-a in Keszthely basin 1976-77

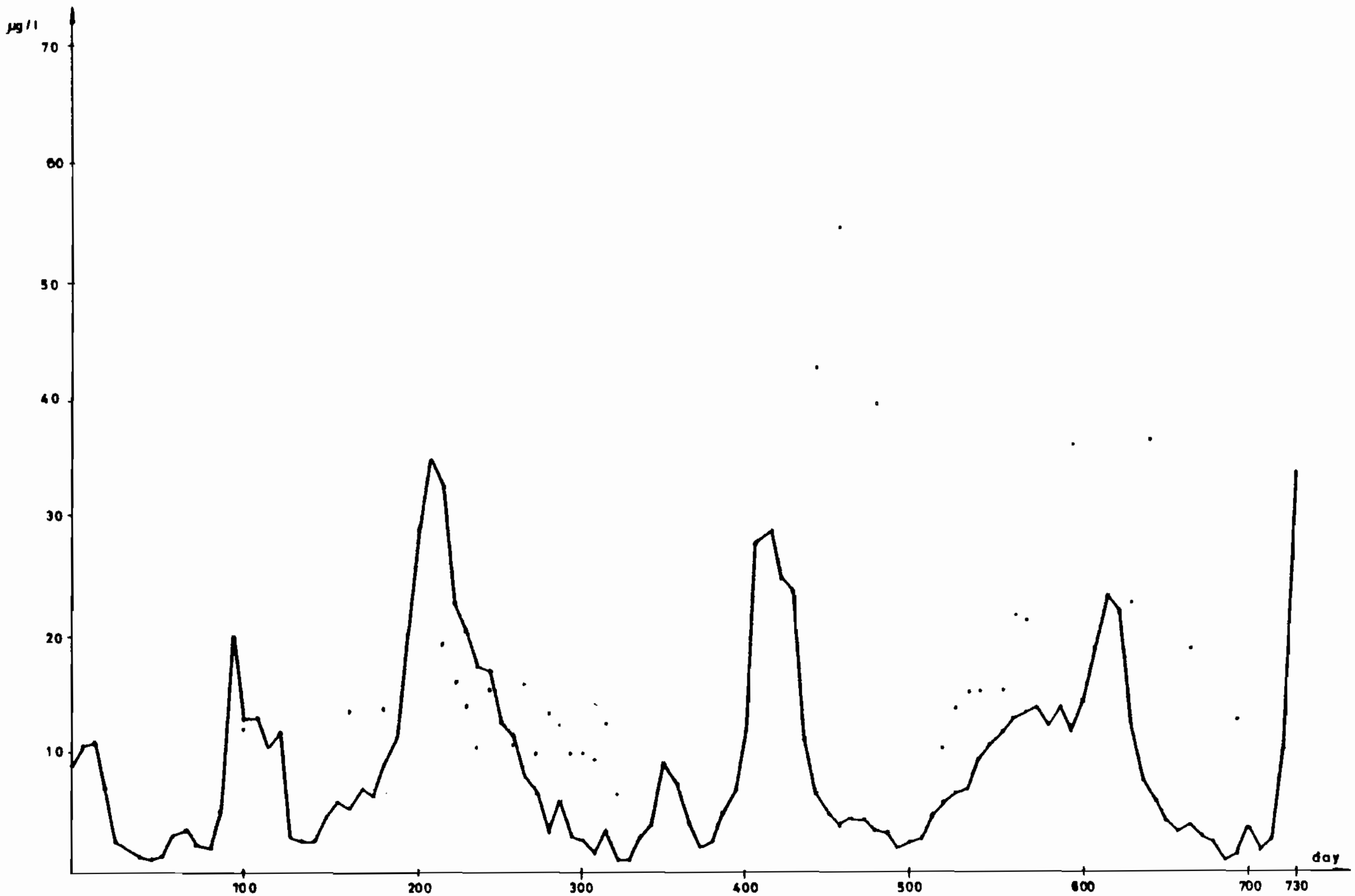

Fig. 3. Measured and simulated chlorophyll-a in Szigliget basin 1976-77

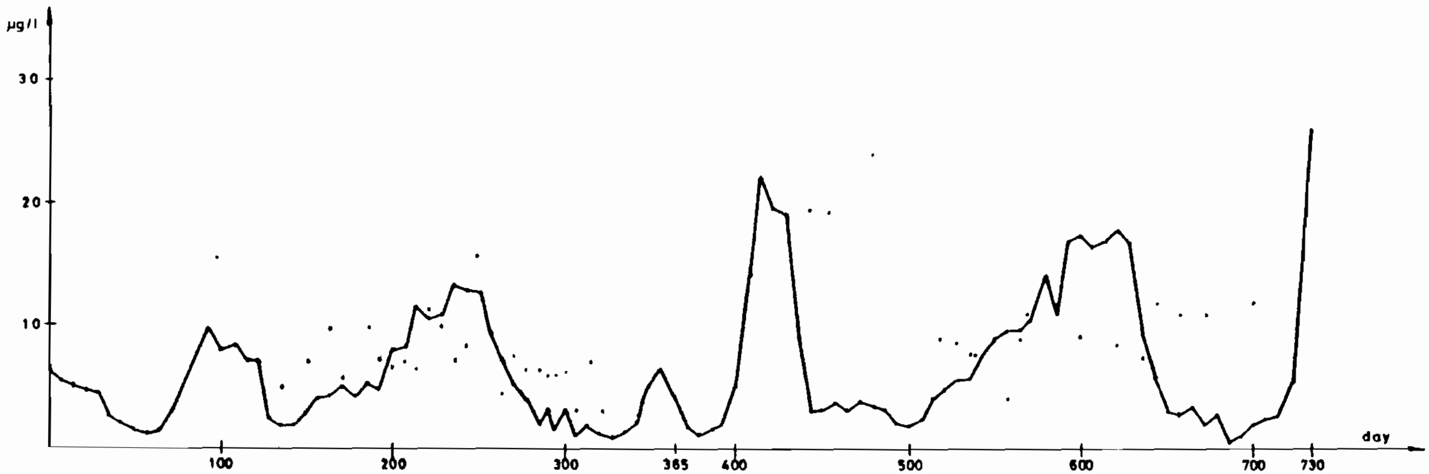

Fig.4 Measured and simulated chlorophyll-a in Szemes basin 1976-77

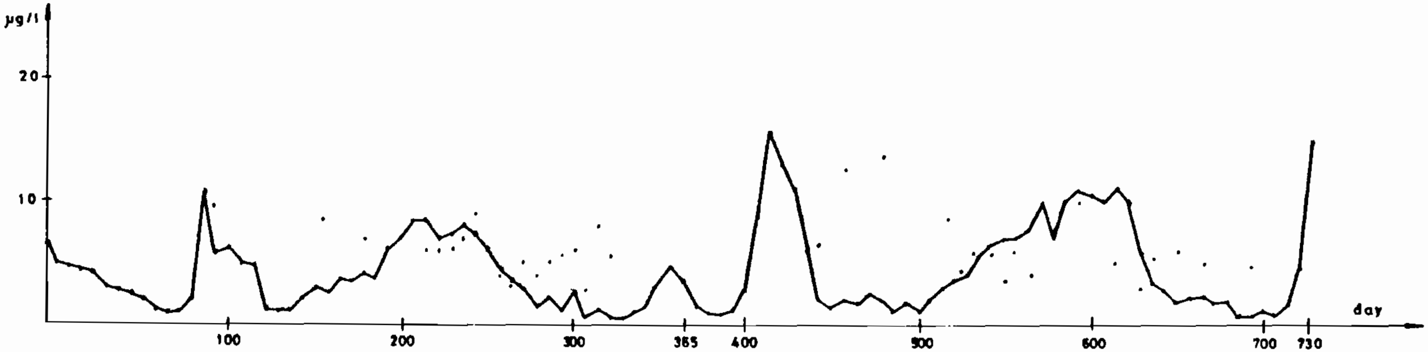

Fig.5. Measured and simulated chlorophyll-a in Siófok basin 1976-77

LOAD REDUCTION SIMULATIONS

As a starting point in dialogue with management modellers and makers some simulation runs were made forecasting the change of water quality due to some reductions of nutrient load.

The phosphorus load has six sources: atmosphere pollution, sewage phosphorus, orthophosphate and particulate phosphorus coming from the watershed, urban run-off orthophosphate and particulate phosphorus. The nitrogen input has two sources: sewage nitrogen and nitrogen coming from the water-shed. The estimation of load is based upon the daily total phosphorus and total nitrogen and the weekly orthophoshpate measurements at Fenékpuszta on River Zala, and occasional measurements on other tributaries and sewage plant outflows.

There are a lot of possibilities to reduce nutrient load, this is the reason why this task needs close cooperation with management makers. As a starting point the orthophosphate coming on River Zala has been decreased to 75%, 50%, 25% and zero. The meteorological data and Zala input data of 1976 were chosen for forcing functions of the model. The model was run for six years (the time series of 1976 was multiplied six times), as six years long simulation can show the tendencies and doesn't need enormus computer time.

Table 2. Summer average phytoplankton biomass in load reduction simulations.

$PO_4=1.0$

basin \ year	1	2	3	4	5	6
1	1.17	1.25	1.24	1.29	1.34	1.37
2	0.61	0.73	0.72	0.75	0.76	0.77
3	0.39	0.38	0.36	0.36	0.35	0.35
4	0.31	0.27	0.27	0.27	0.26	0.26

$PO_4=0.75$

basin \ year	1	2	3	4	5	6
1	1.05	1.06	1.03	1.05	1.07	1.09
2	0.63	0.76	0.75	0.77	0.79	0.81
3	0.39	0.39	0.37	0.37	0.36	0.36
4	0.31	0.27	0.27	0.27	0.26	0.26

$PO_4=0.5$

basin \ year	1	2	3	4	5	6
1	0.93	0.87	0.81	0.81	0.81	0.81
2	0.65	0.76	0.71	0.70	0.70	0.70
3	0.39	0.40	0.38	0.40	0.40	0.40
4	0.31	0.27	0.27	0.27	0.26	0.26

$PO_4=0.25$

basin \ year	1	2	3	4	5	6
1	0.81	0.69	0.60	0.57	0.55	0.54
2	0.67	0.73	0.64	0.64	0.65	0.64
3	0.39	0.41	0.41	0.44	0.46	0.47
4	0.31	0.28	0.27	0.27	0.27	0.27

$PO_4=0.0$

basin \ year	1	2	3	4	5	6
1	0.69	0.50	0.39	0.33	0.29	0.27
2	0.68	0.68	0.61	0.60	0.60	0.60
3	0.39	0.42	0.44	0.48	0.52	0.54
4	0.31	0.28	0.27	0.27	0.27	0.27

As it is impossible to look over and compare the output of six years long simulations, six indicators had been chosen for all simulated years: yearly average phytoplankton biomass, yearly average primary production, summer average algal biomass, summer average primary production, summer maximal phytoplankton biomass and summer maximal primary production. All the six indices show that the reduction of phosphorus load to 50% results only 25% improvement in water quality, e.g. the summer average phytoplankton biomass drops down from 1.17 to 0.81 (see Table 2.). The higher the load reduction the later the steady state in reached, e.g. in case of 50% reduction this period is 3 years while in case of 75% reduction is 6 years.

An interesting and very important question is: what is the effect of Zala load reduction for the other basins. In long term of course the water quality improves, but in the case of decreasing the input of Keszthely basin the water quality of Szigliget basin for a short period becomes worse later improves, e.g. the summer average phytoplankton biomass in Szigliget basin is 0.61 in the first and 0.77 in the sixth year without changing the nutrient load, these values at 75% reduction are 0.67 and 0.64 respectively. This fact can be seen in the case of other consecutive basins too.

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COMPARISON OF WATER QUALITY MODELS OF KESZTHELY BAY

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Abstract

A number of lake eutrophication models containing different combinations of state variables and degrees of complexity are calibrated using the extensive data base available for Keszthely Bay, Lake Balaton, Hungary. Calibration to 1977 data results in models which fail to verify using 1976 data. The calibrated models are then tested for the response of their algal predictions to reductions in PO4 loads discharged by the Zala River during 1977. Large differences in model load responses are found.

INTRODUCTION

Recently, lake water quality management efforts have attempted to balance the dual need to protect and use lakes as water resources, based on the premise that both protection and use are compatable. To do this lake managers must somehow quantify the cause and effect relationships between watershed-based human activities and possible lake water uses. As a result of recent environmental legislation, many countries now require lake management agencies to determine lake water use benefits resulting from alternative pollution control strategies and to balance the costs and benefits prior to control plan implementation. This legal constraint on managers has created a need for the development of mathematical computer models capable of forecasting lake trophic states lake trophic states under various control scenarios.

STUDY OBJECTIVES

Lake managers in need of an answer to the question of lake response to various control measures often rely heavily on existing eutrophication models. However, there is an unfortunate tendency on the part of lake managers to apply a eutrophication model as an expedient predictive tool with very little consideration of its applicability to the lake eutrophication problem under study.

In recognition of this problem, the study objectives are:

1. To demonstrate that a number of eutrophication models containing different combinations of state variables and having varying degrees of complexity can be calibrated using a given set of lake water quality data.

2. To demonstrate that successful calibration does not necessarily result in eutrophication models that can be verified using lake data for for other years.

3. To demonstrate that various lake eutrophication models predict quite different responses to the same nutrient load reductions.

In order to carry out these objectives five eutrophication models of varying complexity have been developed and tested using the extensive data base available for Keszthely Bay, Lake Balaton, Hungary (Baker, 1982). Four of these will be described below. Each model has been incorporated as an optional reactive subroutine within a linked hydrodynamic/biogeochemical eutrophication model of Lake Balaton (Shanahan, 1981).

LAKE BALATON BACKGROUND

Lake Balaton is the largest lake in central Europe, with a length of 78 kilometers, an average width of 8 kilometers and a surface area of 600 km. In recent years a steady lakewide progression towards eutrophic conditions has been observed (Herodek, 1977). Primary production measurements made since 1972 have indicated a trend toward eutrophy in the lake, and annual primary production estimates for Lake Balaton, examined by van Straten et al. (1979), resulted in eutrophic state classification for all areas of the lake. Most notably, Keszthely Bay fell on the extreme hypereutrophic end of the scales.

A similar conclusion is reached from empirical models for lake trophic state classification (Vollenweider (1968) and Dillon and Rigler (1974)).

THE LAKE BALATON DATA BASE

The long-established significance of Lake Balaton as a water resource in Hungary has resulted in the development of an extensive data base, including meteorological and hydrological observations recorded more or less continuously for more than 50 years. An excellent review of the entire data base has been given in van Straten et al. (1979).

IN-LAKE WATER QUALITY PARAMETERS

Detailed data on the spatial and temporal variations of in-lake water quality parameters is vital to the development and application of lake eutrophication models. It is important, not only to have data which is analogous to biological compartments, but that the data be obtained with enough frequency to allow correspondence with time scales of variations within compartments (Jørgensen, 1980). Thus, if an algal population blooms and dies off within a 1-month period, biomass or chlorophyll-a data obtained every 2 months will not be able to capture bloom dynamics.

The Keszthely Bay data for water quality parameters consists of approximately 8 measurements per year at each of 4 stations within the bay. Directly measured parameters include: chlorophyll-a, phosphorus constituents and nitrogen forms. The time-course of chlorophyll-a observations exhibit large variations between years. In the summer of 1975, peak chlorophyll-a concentrations in Keszthely Bay reached almost 70 μg/ℓ. Yet, during the next year, data suggests that no explosive algal blooms occurred. Then, in the summer of 1977, intense algal blooms occurred similar to those in 1975. These highly dynamic year-to-year variations in algal bloom development are well correlated with the dynamics of the Zala River PO4 load, as blooms observed during the summer months in 1975 and 1977 can be seen to be a quick response to spikes of PO4 discharged from the Zala River. The absence of serious blooms during the summer of 1976 corresponds to an exceptionally low loading of PO4 from the Zala River (c.f., van Straten et al., 1979). This rapid response of algal dynamics to changes of the external PO4 load suggests that the role of bottom sediments in short-term PO4 recycling within the bay may be of reduced importance. However, sediment recycling of PO4 may be important in determining the long-term trophic status of Keszthely Bay (van Straten, 1980).

MODEL DEVELOPMENT

As a first step the problem was bounded in space, time and subsystem to include only summer phytoplankton blooms occurring in Keszthely Bay. This corresponded to the most pronounced area of eutrophication in the lake over a time span when data was most complete and algal blooms most significant.

The water quality data available for Keszthely Bay suggests that it is well mixed vertically and horizontally. Furthermore, a time/length scale analysis conducted by Baker (1982) has shown that hydrologic through-flow and longitudinal mixing dominate over transient hydrodynamic effects in influencing summer algal blooms. Therefore, characterization of transport and mixing processes within the bay may be accomplished in an inexpensive but rigorous manner through the use of a 1-box transport component. This approach is used with each water quality model of this study.

Existing models of Lake Balaton are of varying complexity, e.g., BALSECT (Leonov, 1980), BEM (Csáki and Kutas, 1980) and SIMBAL (van Straten, 1980). Of these SIMBAL is the least complex containing 4 state variables and 22 parameters. Each model contains similar formulations for processes controlling algal bloom dynamics, e.g., growth, mortality; however SIMBAL offers, most notably, a simplified sediment interaction system. Balancing the lack of data available to calibrate sediment exchange parameters with the importance of maintaining some sediment interaction for eventual model response testing to PO4 load reduction, a modified version of SIMBAL was included as Model II.

The following is an overview of each of the four water quality models used herein. Further detail including governing equations, etc., can be found in Baker (1982).

MODEL I

This model is similar to SIMBAL as it contains PO4, algal phosphorus and detrital phosphorus compartments. However, only algal and detrital phosphorus settling are included to account for sediment water interactions. A schematic of the biogeochemical compartments is given in Figure 1. SIMBAL's temperature and nutrient limitations of algal growth are used, but the light limitation formulation was modified to eliminate photoinhibition effects which apparently do not occur in Keszthely Bay. In addition, algal phosphorus is loaded into the phytoplankton phosphorus compartment and allowed to settle out at a constant rate.

Model II

This model is the modified version of SIMBAL. All of SIMBAL's state variables and processes are retained. However, algal phosphorus is loaded into the summer phytoplankton phosphorus compartment and then allowed to settle out at a constant rate. In addition, the algal growth limitation by light is modified to exclude photoinhibition effects.

Model III

This model is a modified version of a variable cell quota phosphorus cycle model developed by Wang (1981). The model structure is shown in Figure 3. Algal growth is modeled as a two-step process involving separate PO4 uptake and cell growth mechanisms.

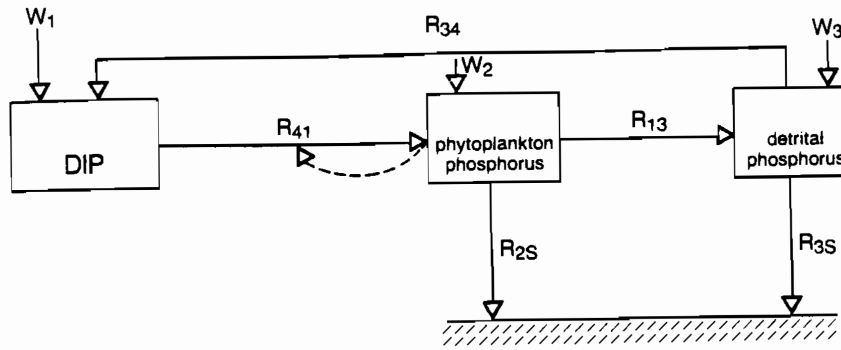

Figure 1. Model I - 3 state variables, 15 parameters; DIP - dissolved inorganic phosphorus

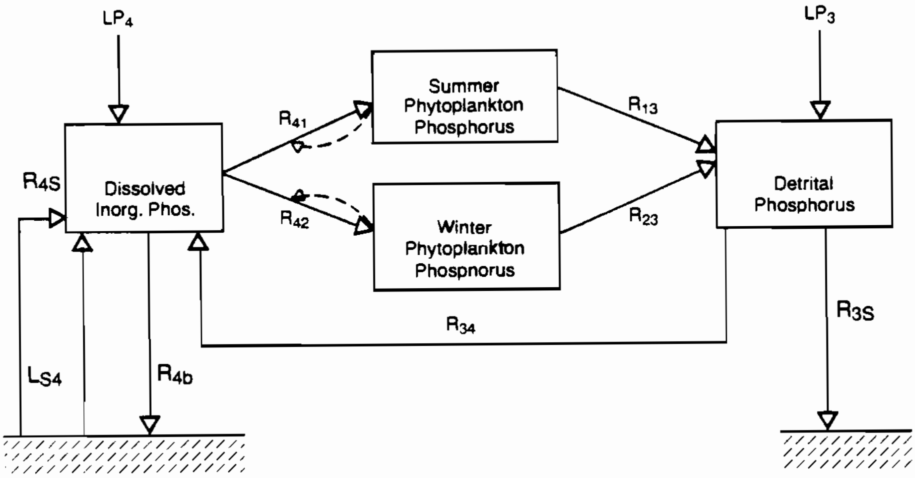

Figure 2. Model II - (SIMBAL) 4 state variables, 22 parameters

The algal biomass growth rate is controlled by light, temperature and phosphorus cell quota using a multiplicative limitation function. Mortality of algal biomass is modeled using the same temperature dependency as Models I and II. Algal phosphorus growth, mortality and settling rates are calculated at each reactive iteration by multiplying the analogous biomass process rates by the cell quota, Q. Algal

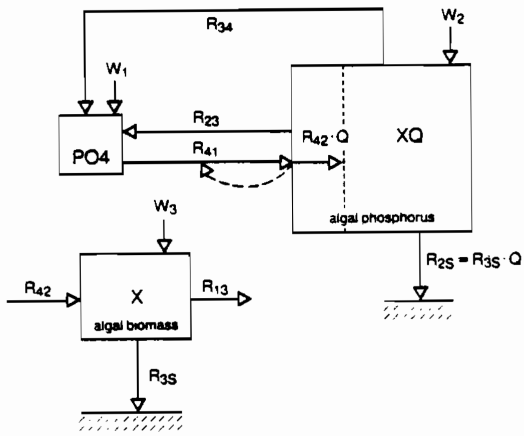

Figure 3. Model III - 3 state variables, 15 parameters

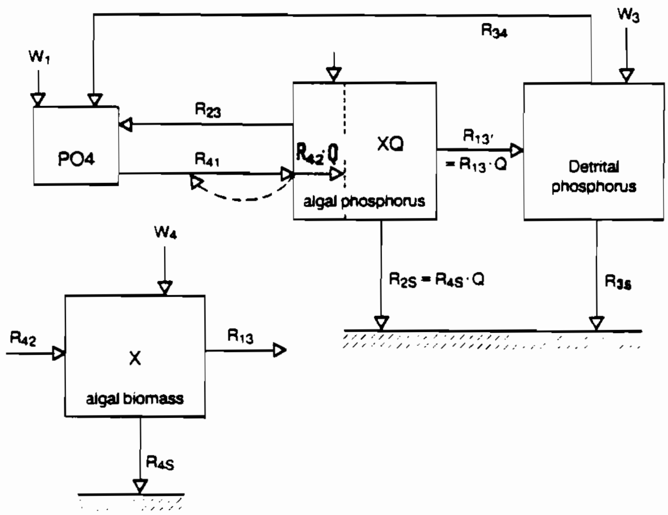

Figure 4. Model IV - 4 state variables, 18 parameters

phosphorus and biomass are lost to the sediments using the same settling rate formulation as Models I and II.

All three compartments receive external loads from the Zala River on a daily specified basis. Since PO4 concentrations in the Zala River are always high (≅ 40 μg/ℓ) it is assumed that algae discharged by this river are at their maximum phosphorus cell quota.

Model IV

This model is similar to Model III; however, it includes a detrital phosphorus compartment. The detritus acts as a storage for the phosphorus in dead algal cells for the purpose of simulating morre realistically the PO4 levels within the water column. A schematic diagram is given in Figure 4.

CALIBRATION AND VERIFICATION OF EUTROPHICATION MODELS

The models of this study are first calibrated via trial and error to Keszthely Bay data for June 13 through November 22, 1977. Following calibration an attempt is made at model verification using data for June 13 through November 11, 1976.

Comparison of Figures 1-4 shows that the four models have a number state variables, and processes in common. Calibration was greatly simplified by holding common parameters constant and only varying those controlling algal compartments. This reduced the number of parameters varied for calibration to 7, 7, 7 and 8, in Models I - IV, respectively.

Calibration results are found in Figure 5. It is seen that all the fixed cell quota models successfully captured seasonal changes observed for chlorophyll-a, but failed to predict short-term algal bloom dynamics. However, the variable cell quota models had the ability to capture both short and long-term algal biomass dynamics.

During model verifications all model parameters were held at calibrated values. However, 1976 initial conditions, hydrologic through flows, forcing functions (solar radiation and water temperature) and external loads were input to the models. The verification results are shown in Figure 6. Due to the monthly basis of observations, the verification is open to question; however, some observations on the success and failure of model verifications can be made. It is seen that Models I

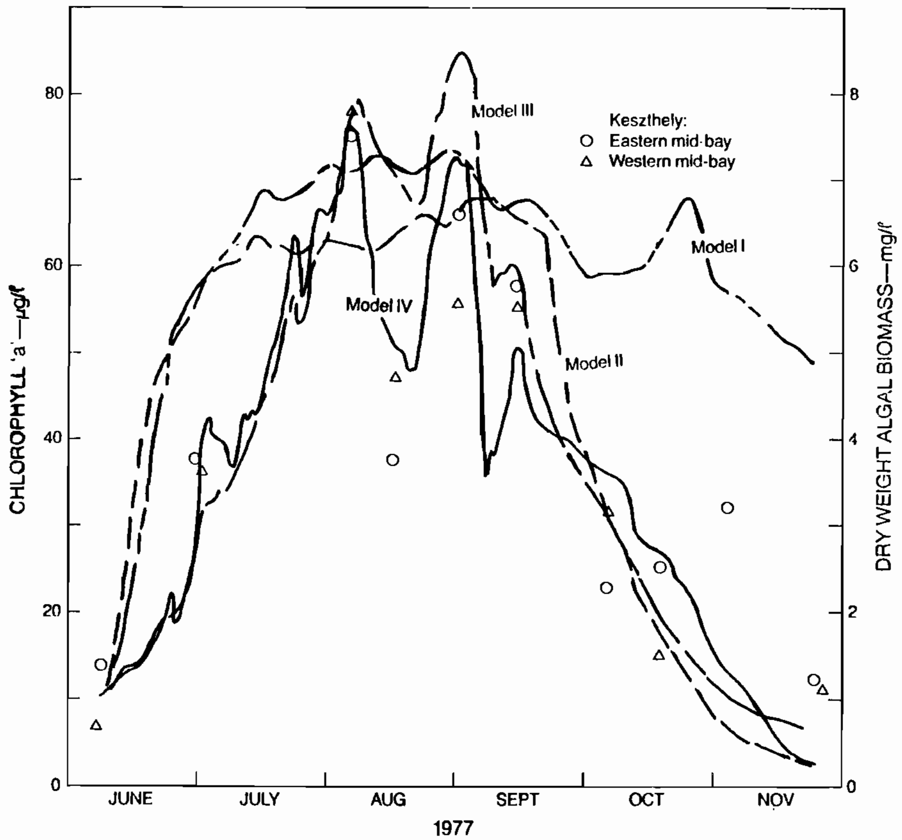

Figure 5. Model Calibration (1977) Results

Ratio of chl-a to d.w. biomass assumed to be 0.01

and II fail to capture even general observed levels of chlorophypll-a. Model III simulates chlorophyll-a reasonably well during June and July, but fails to predict observed levels during the period of August through November. Model IV predicts June, July, September and October levels adequately, but overpredicts chlorophyll-a during August and underpredicts it during November.

From the above observations it is concluded that none of the four models exhibit a satisfactory verification for 1976 conditions. However, the variable cell quota models (III and IV) are more able to capture trends in algal abundance than fixed cell quota models (I, and II).

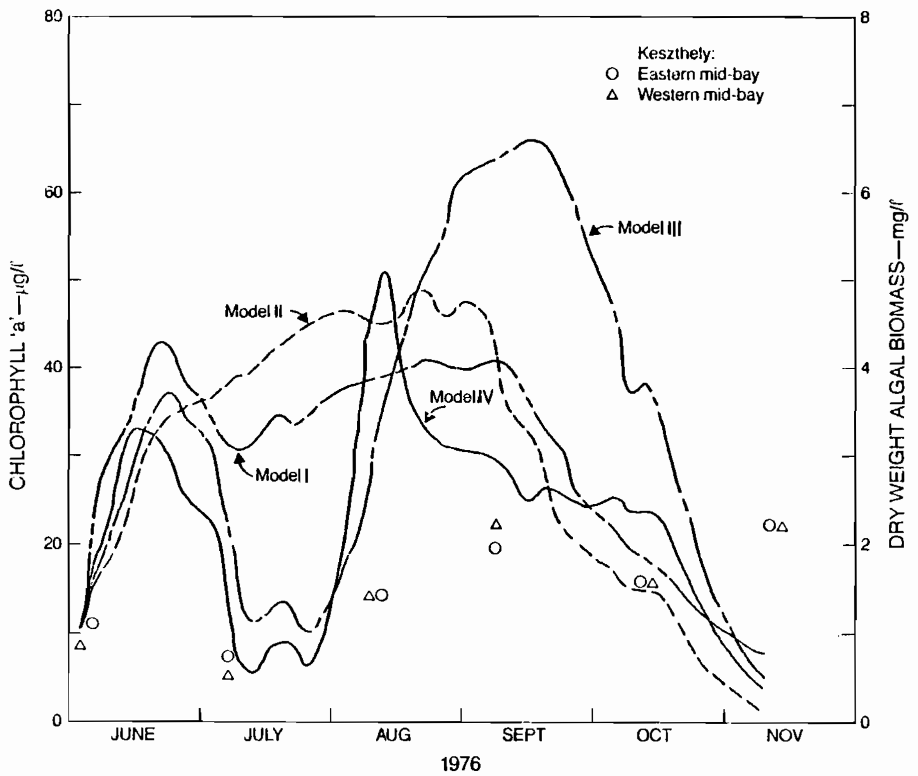

Figure 6. Model Verification (1976) Results.

Ratio of chl-a to d.w. biomass assumed to be 0.01

MODEL LOAD RESPONSES

The daily Zala River PO4 loads input to the models during calibration were attenuated by 10, 30 and 50% and each model was executed using these loads and 1977 environmental conditions. Results of the load response runs are summarized in Figure 7, which shows the response of the summer peak algal phosphorus predictions of each model. It is seen that the models respond quite differently to Zala River PO4 load reductions.

RESULTS AND CONCLUSIONS

In this study it is demonstrated that four eutrophication models of different complexity and state variable combinations can be cali-

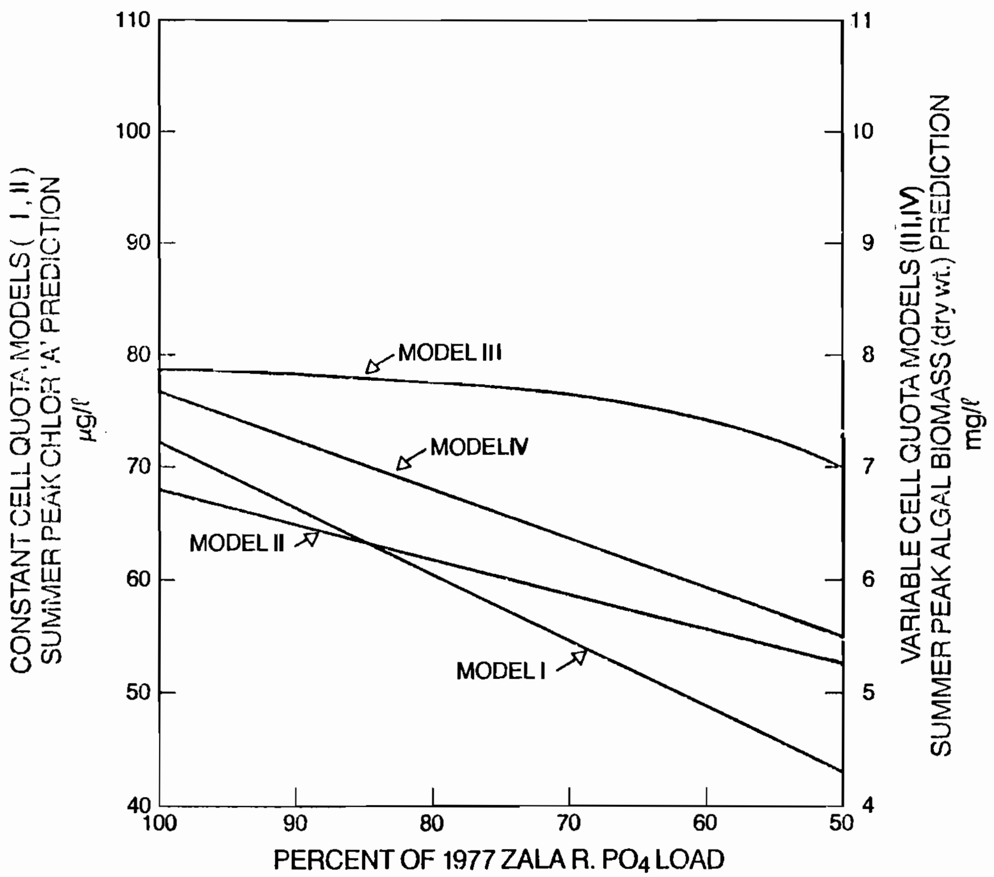

Figure 7. Response of Model Peak Summer Algal Biomass Predictions to Zala River PO4 Load Reduction. Ratio of chl-a to d.w. biomass assumed to be 0.01.

brated to Keszthely Bay water quality observations for a given set of environmental conditions and PO4 loads. However, successful calibration results in parameter sets which are not applicable to the simulation of other time periods where environmental conditions and PO4 loads loads are different. Thus, calibrated models should not be interpreted as valid abstractions of lake eutrophication processes until their ability to simulate changing environmental conditions is verified. Verification under changing environmental conditions is absolutely necessary if the models are to be used to predict responses to control of external PO4 loads. The calibrated eutrophication models of this study exhibit a wide range of responses to PO4 load reduction. Thus, use of a particular calibrated model for assessment of alternative load controls is open to question.

RECOMMENDATIONS FOR FUTURE RESEARCH

Future research should focus on the development of better procedures for data collection and eutrophication model calibration. One way to fulfill this need is through the use of experimental lake enclosures. Mixing within the captured water column (the in situ lake enclosure) can be artificailly controlled in order to study the effect of turbulence and water column circulation on plankton settling rates and light limitation or inhibition. An enclosure may include or exclude lake bottom sediments, thus providing an opportunity, in combination with controlled mixing, to provide valuable information on sediment-water column interactions. The rigorous analysis, by means of mathematical models, of observations made under the controlled conditions facilitated by the enclosure, should provide new insights and confidence in our ability to simulate biogeochemical processes.

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A COMPARISON BETWEEN THE LAKE ERIE AND LAKE BALATON WATER QUALITY STUDIES

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ABSTRACT

Both Lake Erie and Lake Balaton are subject to man-made eutrophication. The two systems show many similarities: shallowness, elongated shape, the distinction of basins different in character, the nutrient loading figures, the prominent longitudinal gradient of various water quality components, the dominant influence of climatic factors, the water circulation patterns, the wind induced sediment-water interaction etc. In addition, the approaches adopted for studying the two systems also show apparent similarities. A preliminary comparison is given between the two systems through various experimental and simulation results ranging from the in-lake processes to the problem of water quality management.

1. INTRODUCTION

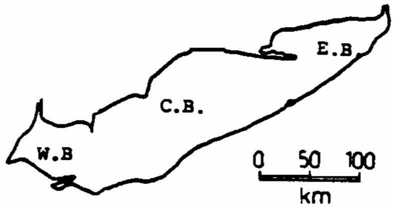
Very often, water quality models are designed specifically for a particular lake. Occasionally, the models or the submodels may be adapted to other lakes by re-adjusting the coefficients. In general, the question of the portability of models often arises, as well as much broader questions. For example, how does a modeller normally adopt a modelling approach for a given set of management objectives and available scientific information? The spatial and temporal details are certainly the main concern in formulating the approach; so are the ecological details. The choice of methodologies appropriate for simulating these details is the next question. The recent advancement in such areas as parameter estimation, system identification and optimization procedures has removed much of the guess work in achieving a good fit with observed data. The modeller must therefore reconcile the use of these powerful modelling tools with the limitations imposed by the data uncertainties and the model complexity. Following the calibration, validation, verification and sensitivity analysis of the model, when the modeller is about to table the results, he faces an even tougher decision. Is the model any good for management application, or should he concentrate all the time on refining the model? This paper is an attempt to address these questions by using examples from the water quality modelling studies of Lake Erie and Lake Balaton with which the authors are familar. It is clear that the questions are of concern to other similar studies, and we feel that an intercomparison of the two modelling studies is unique and timely in terms of providing some of our mutual experiences and answers.

2. MAJOR FEATURES OF THE TWO SYSTEMS

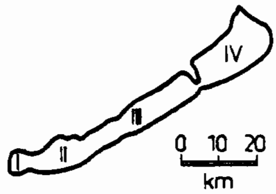
Lake Erie is the shallowest of the North American Great Lakes, consisting of the Western, Central and Eastern Basins. In many respects (see Table 1), Lake Erie resembles Lake Balaton. In the late 1960's, Lake Erie was eutrophic. A phosphorus removal program has been implemented as a result of the 1972 U.S.-Canada Great Lakes Water Quality Agreement. Since then, the water quality of the lake has improved; Lake Erie is now mesotrophic (Fig. 1). The Lake Erie water quality modelling study (Lam *et al.*, 1983) at the National Water Research Institute, Canada, aims at developing computer models based on the lake surveillance data collected during the period of 1967 to 1978. Typically, there are four to six observational cruises per year. A very noticeable feature of the water quality data, in the Western Basin, e.g. total phosphorus concentration, is the presence of irregular and pulse-like concentration maxima caused by the wind-wave resuspension of sediments. Another important climatic influence on Lake Erie Basins is the seasonal thermal stratification cycle. In the Central Basin, the hypolimnion sometimes becomes anoxic and there is chemical regeneration of phosphorus from the sediment (Lam *et al.*, 1982).

Table 1. Comparison of Lake Erie and Lake Balaton

Lake Erie



Lake Balaton



	Lake Erie				Lake Balaton				
	W.B.	C.B.	E.B.	Whole Lake	I	II	III	IV	Whole Lake
mean depths (m)	6.5	17.6	25.9	18.7	2.3	2.9	3.2	3.7	3.1
maximum depths (m)				65					11.6
area (km^2)				25320					596
residence time (y)	0.1	1.6	0.8	2.4	0.3	0.7	1.0	1.3	2.0
TP load ($mgm^{-3}d^{-1}$)	1.5	0.13	0.1	0.09	3.2	0.55	0.31	0.24	0.45
TP conc. (mg/m^3)	36	27	18		80	60	40	30	
max. Chl-*a* conc. (mg/m^3)	24	12	7		75	45	30	15	
climatic influences	thermal stratification; 3-month ice cover; strong westerly wind in winter				no stratification; 2-month ice cover; strong wind action				
hydrodynamics	west to east hydraulic flow dominates; gyres; wind waves				fast dynamics; seiche; gyres; shallow waves				
chemistry	anoxic chemical regeneration of phosphorus and ammonium; oxygen depletion in C.B. hypolimnion				high alkalinity and pH (8.5) oxygen saturation; biogenic lime precipitation				
biology	mesotrophic; phosphorus limiting; spring diatom bloom; autumn algal peak Chl. *a* conc. steady				hyper-eutrophic; spring diatom bloom; autumn algal peak: Chl. *a* increasing conc.				
sediments	till clay and mud; wind-wave resuspension; anoxia due to sediment oxygen demand				fine sand; poor in organic material; physical resuspension; some aerobic regimes				
data	extensive weather and water quality data; surveillance program (4-6 cruises per year)				long hydrological and weather records; frequent water quality survey since 1971				
management	international agreements for phosphorus cleanup (1972, 1978)				phosphorus control is being under realization				

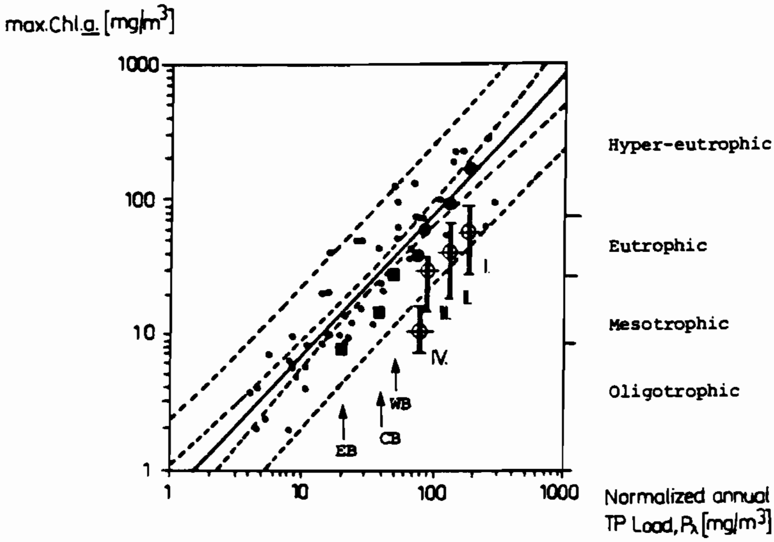


FIG. 1 Trophic levels of Lake Erie, Lake Balaton and OECD lakes.

■ Lake Erie basins (1975-1978 data): WB- Western Basin
CB- Central Basin
EB- Eastern Basin

Lake Balaton basins (1975-1979 data): I- Keszthely Basin
II- Szigliget Basin
III- Szemes Basin
IV- Siofok Basin

● Lake Balaton basins (1982 maximum values)

• OECD lakes

By contrast, Lake Balaton does not have a comprehensive loading reduction program and is rapidly becoming hyper-eutrophic (Fig. 1). While it does not have stratification problems, it is strongly affected by wind-wave actions as Lake Erie. Thus, the modelling approach of the Lake Balaton study (Somlyódy and van Straten, 1983) must also take into consideration of weather influences, sediment return, hydrodynamics, nutrient loading, biological and chemical kinetics, in much the same way as the Lake Erie study.

Figure 2a shows the total phosphorus loadings to Lake Erie for the period of 1967 to 1978. There is a marked decrease in the order of 50% over the period. For example, the Detroit River source, representing an average of 56% of the total loading, drops from 14309 to 6205 MT per year over the twelve-year period. Similar reductions of approximately 50% are noted for the municipal and industrial sources. In general, the decline in loading is due to the implementation of the U.S.-Canada Great Lakes Water Quality Agreements of 1972. On the average, the loading of total phosphorus to the Western Basin is about 1.5 mg m^{-3} day^{-1} as compared to 3.2 mg m^{-3} day^{-1} for the Keszthely Basin of Lake Balaton. The relative contribution of the Lake Balaton total P load is about 47% from agriculture and 28% from sewage, and the available P load is 33% from agriculture and 52% from sewage; other sources include atmospheric loads (6% and 4%, respectively). These phosphorus loads of Lake Balaton do not show a major decline over the past seven years.

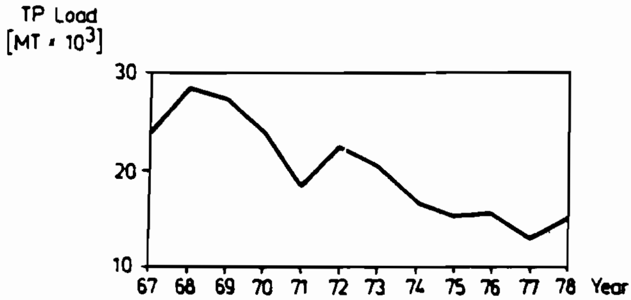


FIG. 2(a) Total Phosphorus Loading to Lake Erie

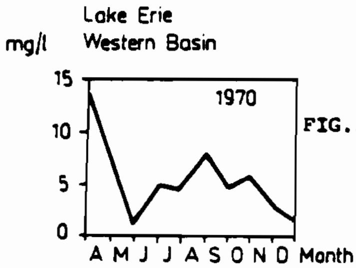
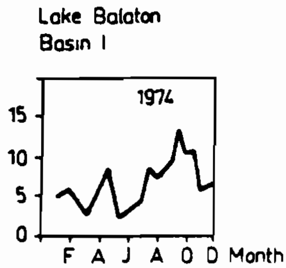


FIG. 2(b) Observed Biomass (mg/l)



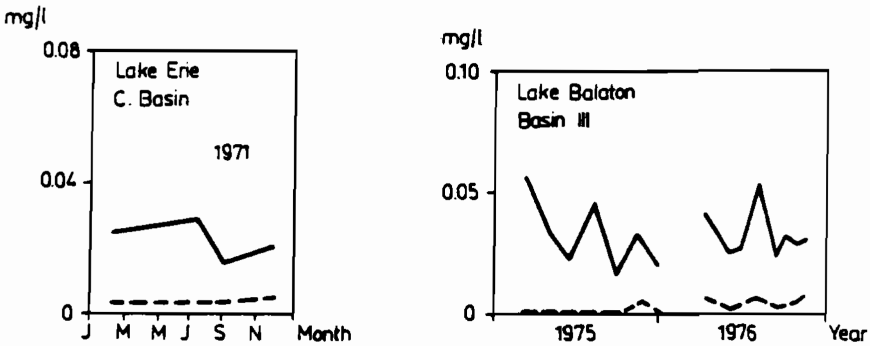


FIG. 2(c) Observed concentrations TP (——) and SRP (----)

Because of the enrichment from nutrient loadings, both lakes support active biological growth. For example, the average biomass in the Western Basin of Lake Erie reached a peak of about 14 mg/L in 1970 and reflects a similar value in the Keszthely Basin in 1974 (Figure 2b). However, for the past few years, while the primary productivity declines steadily in Lake Erie, Lake Balaton shows rapid increases in algal growth. Likewise, the chemistry data reflect the distribution of loading sources. Both lakes show a decrease in the concentration, e.g. total phosphorus concentration along the longitudinal axis of the lake (Table 1), since the major loading source is located at the west end of the lake. Note that the total phosphorus concentrations of Lake Balaton (van Straten et al., 1979) are about twice as large as those of Lake Erie (Lam et al., 1983).

Another remarkable feature of the lake chemistry concerns the relative proportion of soluble reactive phosphorus present in the total phosphorus. Typically, the soluble phosphorus of both lakes stays low, at about 0.001 mg/L throughout the year (see Figure 2c), i.e. about 3 or 5% of the total phosphorus concentration in the case of Lake Erie (1 or 3% in the case of Lake Balaton). The uptake of this soluble form as nutrient by algae probably explains the depletion in the summer, but cannot explain the depletion in the winter. Indeed, the typical winter values of soluble reactive phosphorus and total phosphorus in deep lakes, e.g. Lake Ontario (maximum depth of 230 m) are about 0.015 and 0.025 mg/L respectively, i.e. 60% being soluble (Simons and Lam, 1980). Since Lakes Erie and Balaton are relatively shallow, such persistent depletion of soluble reactive phosphorus is an important feature which warrants attention in shallow lake studies.

3. THE APPROACH

The advantage of using a systems approach to analyse the data and to evaluate alternative management strategies is obvious from the previous discussion on the interactions of nutrient loadings, hydrodynamics, biological and chemical aspects of the lake. There are two extremes of using the systems approach. The one extreme is a drastic simplification of the interactions as used in a one box model of the constantly-stirred reactor type. The other extreme is the use of a large-scale model which accounts for all the subprocesses and species. The appropriate choice is probably intermediate in complexity between these extremes (Lam et al., 1982; Somlyódy, 1982a). The main difficulty in the case of Lake Erie has been the difference in the scales of physical and biochemical phenomena. Whereas one can simulate the total phosphorus concentration in a lake with a mass balance one-box model, the hydrodynamical transport must be obtained by a model with hundreds of cells. One remedy is to decompose the model into submodels, each of which is based on appropriate temporal, spatial and ecological scales (Fig. 3a), so that the information can be aggregated in the overall model for synthesis (Lam et al., 1982).

(a)

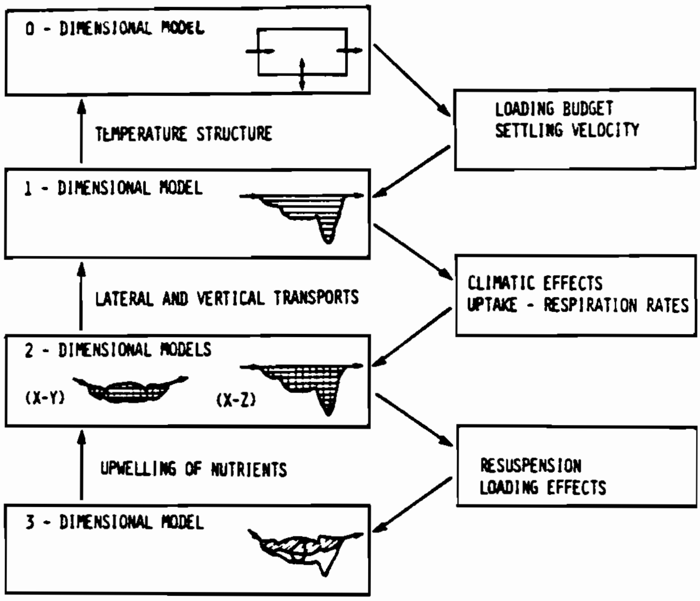
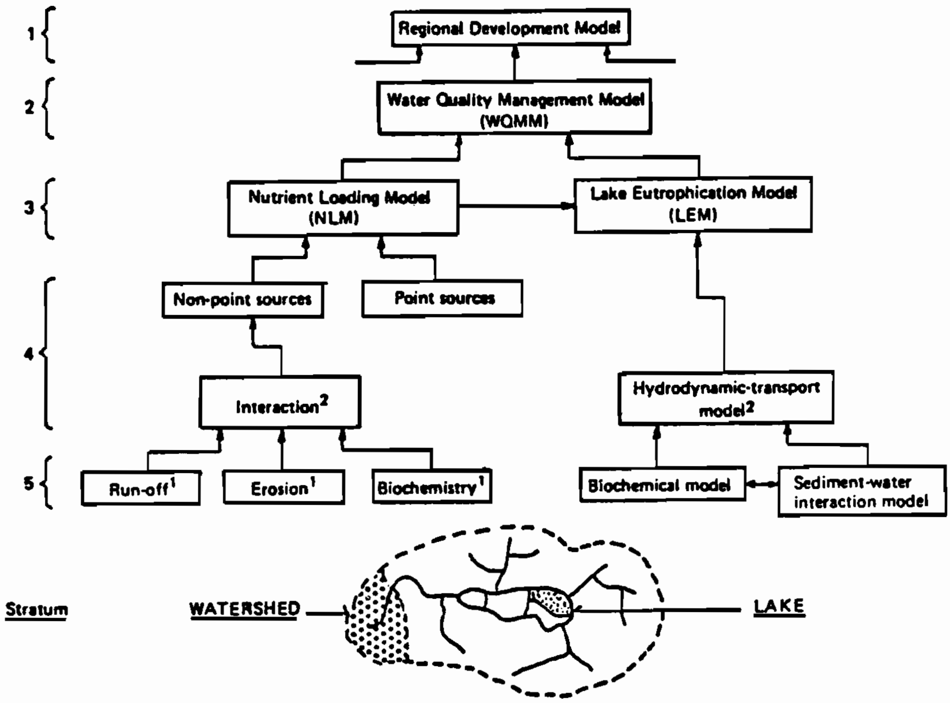


FIG. 3 The modelling approach for (a) Lake Erie and (b) Lake Balaton.

(b)



The approach for Lake Balaton begins by decomposing the system into smaller, tractable units forming a hierachy of issues. Detailed investigations of each of these issues can be based on in situ or laboratory experiments as well as model studies. Only the issues that are important for the higher strata of the hierachy are integrated in the final step. There are five strata (Fig. 3b), ranging from the level of detailed nutrient-plankton submodels to the level of regional development policy model (Somlyódy, 1982a). In principle, such an off-line approach is similar to the Lake Erie modelling approach. Thus, it is interesting to find out if there is some universality in the formation of the hierachy of submodels, or if there are differences in the submodels of the two lake systems. The following are some examples.

4. EXAMPLES

4.1 Hydrodynamics and Transport

The water quality conditions of shallow lakes are more susceptible to wind-induced circulations and waves than those of deep lakes. The hydrodynamic influences are strong in both Lakes Erie and Balaton, although they are somewhat dissimilar. In Lake Erie, the prevailing wind direction is westerly and along the longitudinal axis of the lake. Figure 4a shows a typical record of the currents at Central Basin (Boyce *et al.*, 1980). While there are periods with rapid changes in directions, there are also periods with a persistent easterly or westerly direction for several days. By contrast, the

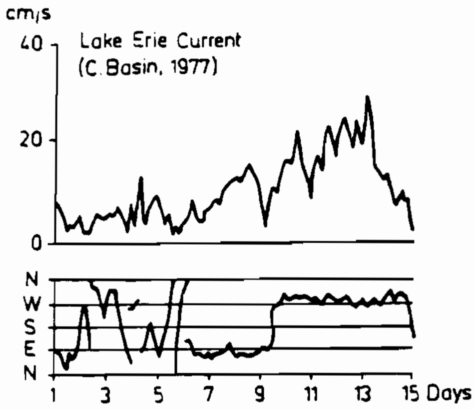


FIG.4(a) Observed Currents (Lake Erie) (velocity and direction)

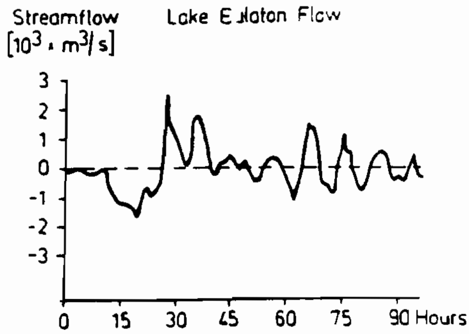


FIG.4(b) Simulated streamflow at the peninsula (Lake Balaton, see Table 1.)

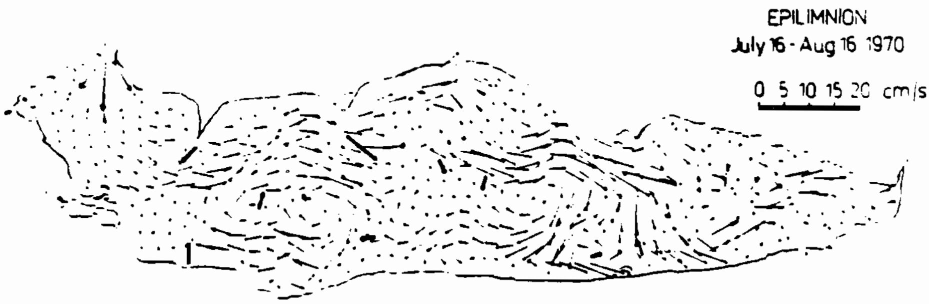


FIG.4(c) Typical circulation pattern for Lake Erie

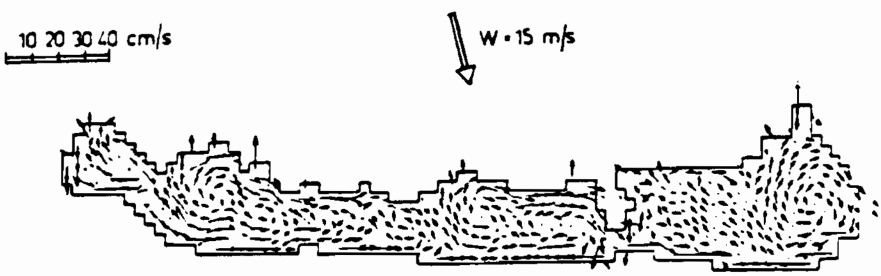


FIG.4(d) Lake Balaton circulation pattern prevailing wind is from the northwest in Lake Balaton, across the width of the lake. The hilly terrains in the north shore also complicate the wind-driven circulation. As a result, there are rapid changes (Fig. 4b) in flow directions (Somlyódy and Virtanen, 1982). Therefore, in terms of characterizing the transport and dispersion of pollutants in Lake Erie, it is possible to use hydrodynamic models (Fig. 4c) to compute the advective flow and to use a constant eddy diffusivity for parameterizing the turbulent diffusion processes (Lam and Simons, 1976). In the case of Lake Balaton, however, a more effective description of such physical influences is to use the fairly constant hydraulic flow and a large-scale, time-varying dispersion component obtained from the computed currents (Fig. 4d) of a hydrodynamic model (Shanahan et al., 1981). In both lakes, the transport and dispersion of water quality variables is a strong function of the meteorology and, to a large extent, dictates the degree of spatial complexity of the water quality model to be chosen (Lam et al., 1982; Shanahan et al., 1981).

Indeed, there has been a growing tendency to reduce the spatial complexity in models, since recent experiences (e.g. Simons and Lam, 1978) showed that detailed, coupled hydrodynamic-transport-water quality models did not greatly improve the predictability and were expensive to run. These experiences have convinced the Lake Balaton researchers to adopt the less detailed multi-segment approach. The selection of the number and sizes of the segments was found to depend on the basin shapes, the circulation patterns and the acceptable level of numerical dispersion (Shanahan and Harleman, 1982). A four-box model is then used with the four boxes coinciding with the four basins of Lake Balaton. Likewise, the Lake Erie study also learned from these previous experiences in adopting a three-box structure. Since the transport and diffusion used in these box models are derived from the aggregation of detailed submodel results (Fig. 3), the box models preserve the essential features of the lake hydrodynamics which are vital for the success in simulating shallow lakes dynamics.

4.2 Wind-wave Resuspension

The other important physical factor affecting the mass balances of nutrients in shallow lakes is the process of wind-wave resuspension. There has been a mutual transfer of modelling knowledge between the Lake Erie and Lake Balaton studies on this subject. Lam and Jaquet (1976) proposed an algorithm (Fig. 5a) for computing the upward flux, J, of total phosphorus from the sediment, based on wave hindcasting methods and grain-size distributions. The empirical formula

$$J = k \, \rho_\omega \frac{\rho_s}{\rho_s - \rho_\omega} \, w_e \tag{1}$$

was found to be appropriate for Lake Erie, where k is constant, ρ_ω is water density, ρ_s is dry sediment density and w_e is excess wave orbital velocity. Somlyódy (1980) modified Eq. (1) by assuming an empirical relationship between w_e and the wind speed, W, so that

$$J = k_1 \rho_\omega \frac{\rho_s}{\rho_s - \rho_\omega} W^m \tag{2}$$

where k and m are constant. Equation (2) can be simplified to

$$J = k_2 W^m \tag{3}$$

and for an optimal fit to Lake Balaton data (Beck and Somlyódy, 1982), m is chosen as 1 and k_2 is determined by the parameter identification procedure with the extended Kalman filter technique (Fig. 5b). This simplification is based on the observation that waves are important only with steady winds during large storms in Lake Balaton and the wind-induced resuspension of the sediment is probably tied in with the turbulent shear associated with rapid changes in wind (Somlyódy, 1980). Figure 5c shows the extended Kalman Filter results for the suspended solids concentration in Lake Balaton. Recently, Lam et al. (1983) used Formula (3) in a simplified Lake Erie phosphorus model for the Western Basin under similar assumptions and the results are quite satisfactory.

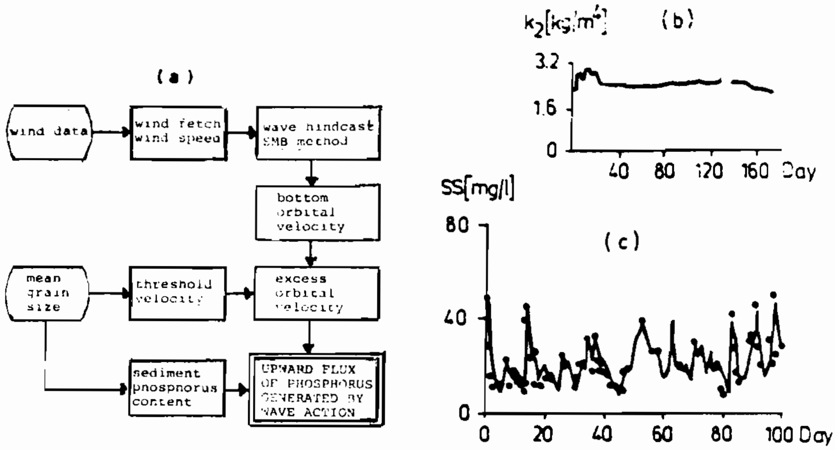


FIG.5 (a) Lake wind-wave resuspension submodel, (b) parameter estimation of k_2 in Eq.(3) and (c) computed (line) and observed (dot) suspended solid concentrations for Lake Balaton.

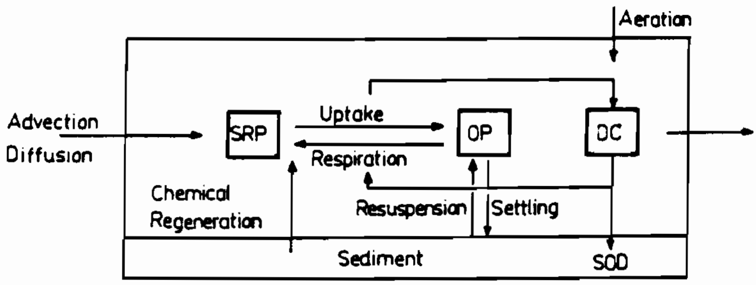


FIG.6 (a) Lake Erie 3-Variable Water Quality Model. (SRP-soluble reactive phosphorus; OP-organic phosphorus; DO-dissolved oxygen)

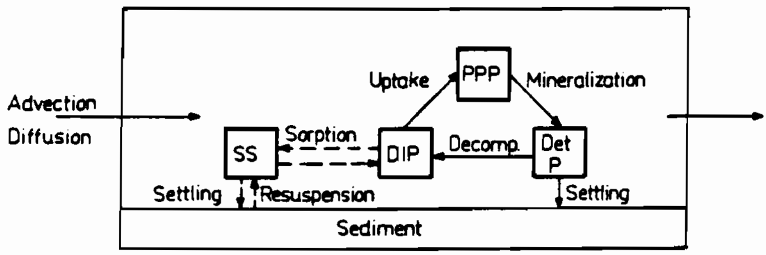


FIG. 6(b) Lake Balaton 3-variable phosphorus model. (PPP-phytoplankton P; DIP-dissolved inorganic P; Det P - detritus P; SS - suspended solids - as implicite state variable)

4.3 Model Structure

There is also a tendency to reduce the ecological details in water quality modelling. In the Lake Erie case, it is found that a three-variable ecological submodel is sufficient (Fig. 6a). In the Lake Balaton case, a number of alternatives have been examined (Somlyódy, 1982a). Figure 6b shows one example (van Straten, 1980) with three explicit state variables of phosphorus and one implicit state variable of suspended solids. The difference in these two types of model structure indicates that, from an ecological point of view at least, there are specific management problems for individual lakes and hence the minimum number of state variables required to address these problems varies. For example, dissolved oxygen is an important water quality variable for Lake Erie but not for Lake Balaton. However, in terms of the phosphorus dynamics, the two lakes require quite similar assumptions on advection, diffusion, settling, algal growth, mineralization, regeneration, and resuspension (Figs. 6a and b). Indeed, the model structures of the two lake systems have similar inadequacy in explaining the low levels of soluble reactive phosphorus as mentioned earlier (Fig. 2c). Preliminary findings for Lake Balaton (Gelencsér _et al_., 1982) showed that the sorption mechanism associated with resuspended sediments could have caused the removal of soluble reactive phosphorus from lake waters. Whether this sorption mechanism occurs in other shallow lakes is still an open question.

4.4 Loading Estimates

Besides the limitations of the limnological knowledge, there are also the uncertainties in the input data which can restrict the application of the model. Basically, there are two main types of input data required by water quality models: the weather data and the loading information. As long as there are operational meteorological stations, there are data available for describing wind, solar radiation and air temperature, etc. Thus, the uncertainties of weather record are not so much tied in with the frequency of sampling or instrument errors but with the large seasonal or even day-to-day fluctuations of the data themselves. These fluctuations can at times be very large and can mask the effects of the loading input.

By contrast, the main difficulty associated with loading estimates is that concentration samples are much fewer than flow samples at inflow sources. Thus, estimation methods are required to minimize these uncertainties. An example is the method by Joláńkai and Somlyódy (1981) which uses a record of detailed streamflow and concentration data to establish the estimation of the load for a given period (e.g. a year). Then, from this data set, suppose only infrequent (e.g. monthly) observations are known. By selecting different infrequent samples, different load estimates are obtained, allowing for a statistical (Monte Carlo) analysis of the estimated data series. As a result, the averaged value, range and variance etc. can be derived and compared to the more accurate estimate so that the error related to infrequent samples can be established. These error statistics gained from such a Monte Carlo analysis can then be extended in a relative sense to other tributaries where only infrequent samples are available. Furthermore, the forcing functions based on the loading estimates can be established in a similar random fashion and the model response to these forcing functions can be obtained using a Monte Carlo simulation. The final results can be viewed, therefore, as if the climatic factors were included (Somlyódy, 1982a).

The Lake Erie water quality modelling study basically used a similar ratio estimator (Fraser and Willson, 1981). However, since the responses to the climatic inputs over a twelve-year span are simulated deterministically (Fig. 3a), there is no need to perform such a Monte Carlo analysis. The analysis may become necessary, however, if weather extremes and stochastic distributions outside the twelve year span are required.

4.5 Seasonal and Long-term Results

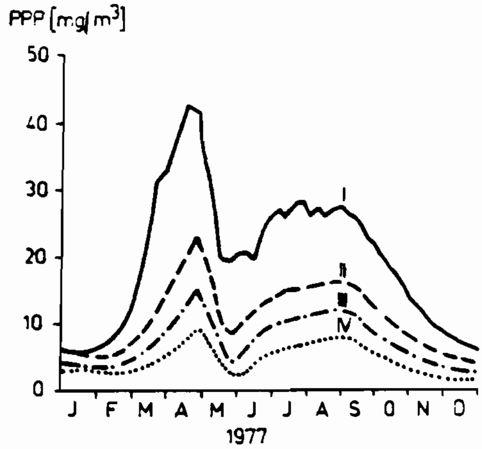
The response of water quality to the weather variations is often short-term and seasonal. One notable example is the water temperature and sunlight which govern the algal growth and decay. Figure 7a shows the responses of algal phosphorus (PPP) to such seasonal variations and other factors (Fig. 6b) for the four basins in Lake Balaton (van Straten, 1980). The marked seasonal changes, particularly the two algal peaks, are characteristics of the ecological system. Under normal weather conditions, this seasonal cycle of algal growth will repeat from year to year. In addition to these regular cycles, however, there are irregular weather episodes such as strong sediment resuspension, which can interrupt the cycle. Thus, when these events are linked together over a long period (e.g. 10 years), the seasonal cycles are not so clearly defined.

An example of long-term results is given in Fig. 7b for the total phosphorus concentration in the Western Basin of Lake Erie. In general, there is a long-term downward trend in both the computed and observation concentrations, corresponding to the phosphorus cleanup program (Fig. 2a). However, such a downward trend is by no means a monotonic decreasing one; there are episodes which show sudden increases in total phosphorus concentration most often as a result of sediment resuspension (internal loads). These irregular phenomena (Fig. 7b), which is attributable to the weather influences, are important considerations in establishing target loads for controlling eutrophication problems in shallow lakes.

4.6 Static Model Responses and their Applications

In spite of the limitations, the models discussed in the previous section simulate the changes in the total phosphorus and the plankton phosphorus quite satisfactorily. By verifying the models with long-term data, these models are actually proven to be capable of establishing useful loading reduction guidelines (e.g. Lam *et al.*,

FIG.7(a) L.Balaton P-model results (SIMBAL, after van Straten, 1980)



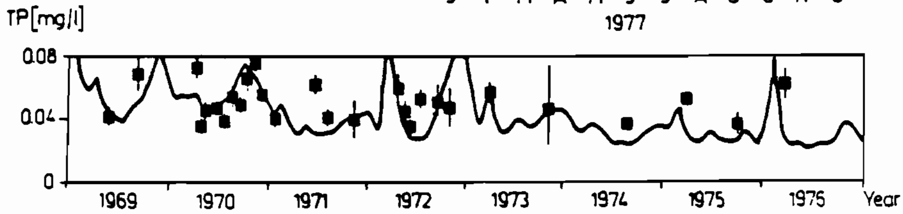
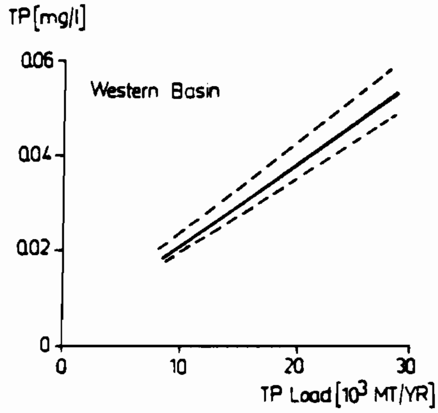


FIG.7(b) Computed and Observed TP Concentrations (Lake Erie, W.Basin)



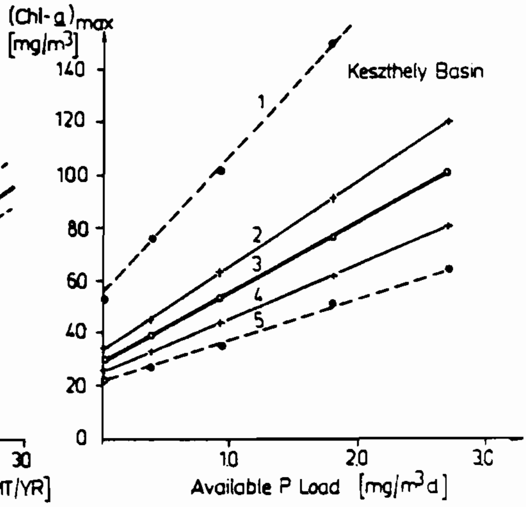


FIG.8(a) L.Erie "static" model results (Lake responses to loading changes under climatic influences)

FIG.8(b) L.Balaton "static" model results (short term response on load reduction derived from the dynamic model) (1 and 5 - extremes from Monte Carlo simulations; 2 and 4 - $\pm$; 3 - mean)

1983; Somlyódy and van Straten, 1983). Figure 8a shows the response curve of the total phosphorus concentration in the Western Basin to changes in total phosphorus loading. Figure 8b shows similar curves for plankton phosphorus concentrations of the Keszthely Basin of Lake Balaton to changes in available phosphorus loading. Both modelling studies indicate that nutrient loading reduction will reduce the phosphorus concentration in the lake and will lessen the problems of eutrophication, and that there is the possibility of internal loads from the sediment, depending on weather conditions.

The curve shown in Fig. 8a is derived from running the dynamic model to steady state under the normal, adverse and favourable weather conditions, all of which are simulated deterministically. The curves shown in Fig. 8b also represent static responses which have been derived by using both deterministic and stochastic methods (Somlyody, 1982b), as discussed earlier in the treatment of loading data (Section 4.4). The application of these response curves is straightforward and should be carefully considered for planning management strategies.

5. CONCLUSIONS

There are probably many more related studies on Lake Erie and Lake Balaton, which are relevant to this intercomparison. However, we feel that there are sufficient results, that we have discussed so far, to arrive at some preliminary conclusions:

(1) The approach of using models with intermediate complexity is appropriate for eutrophication problems of shallow lakes. Consistency in spatial, temporal and ecological details must be sought for during the model development. Our findings ruled out the use of large-scale coupled hydrodynamic-water quality models with hundreds of grid points and over twenty ecological state variables. There is, however, a minimal requirement for the number and sizes of spatial segments as well as the number of ecological state variables for a given set of management objectives.

(2) The model packages are not transferrable without extensive recalibration and modification, although some model formulations are transferrable. Specific submodels are required for individual problems.

(3) Loading and climatic influences are important considerations and their uncertainties must be included in shallow lake modelling studies. The methodologies are well advanced to handle these uncertainties and to provide meaningful results for extrapolation.

(4) Experimental data are needed to verify models before the models can be used to establish management guidelines; the models should be used to identify gaps of knowledge. It is therefore suggested that a mixture of simple and intermediate complex models should be developed to satisfy the research and management requirements. In general, the

dynamic model is associated closely with research and the static model, with management.

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A TWO-STAGE APPROACH FOR LARGE SCALE SEWER SYSTEMS DESIGN WITH APPLICATION TO THE LAKE BALATON RESORT AREA

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Abstract

This paper describes a new, two stage model for sewer system design. In the first stage a large number of so-called main sewer systems are constructed, each of which is consistent from the engineering point of view and none of which is uniformly inferior to any other. In the second stage a pure 0-1 integer programming problem is formulated and solved the basic element of which are the main sewer systems and wastewater treatment plants. The corresponding computer program system functions are outlined. Experiences with the model to the Lake Balaton sewer system planning are summarized. Further research directions, possible applioations and relations to other sewer system planning tools are shortly given.

1. Introduction

From the middle of the 1970's the eutrofioation of the Lake Balaton has been accelerated. At the same time, as a result of advancing urbanization around the Lake, the wastewater load has exceeded the capacities of wastewater treatment plants at several areas and further increase in the quantity of drinking water supply is expected. These facts lead to higher activity both in regional planning and in water quality management. Both of them can use advantageously the results and analysis of a good sewer system design of low cost subject to different side constraints and objectives.

The planning of wastewater collecting systems is generally considered as a routine, straightforward task, but in fact the large number of alternatives, requirements and objectives of different type make the task very complex. This complexity is usually overcome by the long years of experience and engineering standards which results in acceptable solutions from different points of views, but without any assurance of being near the minimal cost solution. Another problem is that the small number of alternatives in the tradional planning does not allow the examination of trade-offs between different objectives and parametric analysis comparing the resources used and the results.

The present practice of the design of wastewater collecting systems essentially consists of three distinct phases:

The first phase, the location of sewer network elements is usually left to the common sense and judgement of the engineer and considered as fixed in the further steps.

The second phase is the estimation of wastewater sources /houshold,- industrial- and agricultural wastewater, stormwater/ and their aggregation according to the predetermined network.

The third phase is engineering calculation of sizes /capacities/ of pipes and wastewater treatment plants in the network.

A number of papers deal with the problems of third and second phase, including the vertical alignement of the pipes in the network: Dajani et al. /1977/, Merrit and Bogan /1973/, Oron /1979/, Walsh and Brown /1973/, Walters and Tempelman /1979/. A special case of the first phase problem is considered by the model of Mays et al./1976/. The lattice type network model is solved by dynamic programming algorithms. The network design problem is also handled by Jarvis et al. /1978/ in which a network flow problem is solved on the prefixed arcs, determining for certain nodes the main question of "treating or transporting" to another wastewater treatment plant.

Our basic problem of sewer system design is substantially different from the ones mentioned above as it was outlined by Kovács et al./1980/. The closest model is the one by Jarvis et al. /1978/. The network flow algorithm is very efficient but it is unable to accept further requirement such as choice of best cost/effectiveness portion, limited resources and limited phosphorus load on the recipient water, parametric analysis of investments, etc. For this purpose in the present paper we shall describe a pure 0-1 discrete programming model.

The second section contains a verbal description of the problem together with a short sketch of the traditional engineering planning. A two stage approach with the definition of the basic elements and the so-called global model is formulated in Section 3. The next section outline the computer program system, which can be used both in batch and in interactive mode for different administrative problem solving and analysing functions. Section 5 summarizes the potential applications of the global model and the program system in management decisions and in the process of engineering planning. Section 6 shows in a condensed form some of the results obtained for the Lake Balaton resort area, while the last section gives the conclusions and some research suggestions.

2. Planning of a sewer system

In the process of tradional engineering planning of sewerage the necessary data are gathered /topographic data, sources of waste water, recipient water etc./, then the existing and potential location/size of main sewers and waste water treatment plants are determined, including all investment and operational costs. Finally a few complete alternative sewer systems are planned by very tiresome hand calculations, one of which is chosen according to certain subjective and objective measures. Although a large part of data preparation can also be aided by computer, in the present paper we mainly restrict ourselves to the actual planning and decision making part.

In the rest of the paper REGION is an area containing one large or several settlements and there are different possibilities of sewering /including local and regional ones/, furthermore the desposing of cleaned waste water is identical /same recipient water, irrigation of a given land etc./

For the purpose of further discussion, now we give a short, simplified verbal description of the so-called global planning problem of sewer systems:

It is given a REGION with its BASIC DATA /all types of data mentioned above/. Determine a subset of potential MAIN SEWERS and of potential WASTE WATER TREATMENT PLANTS and their sizes of MINIMUM TOTAL COST, subject to the following conditions:

- the system is consistent /from the engineering point of view/
- at least a predetermined percentage of the population is sewered
- further special requirements are fulfilled, such as
  - obligatory sewering at certain locations or subregions
  - minimum sizes for certain system components are given,
  - limit on the total outcoming quantity of phosphorus etc.

The expression of system consistencyrefers to the fact, that the laws of nature should be obeyed like in all engineering planning:

- law of oonservation of material
- gravity direotions
- sufficient oapacity for each main sewers /gravity and pressure/
- directed connectedness of all main sewers to a waste water treetment plant of sufficient capacity.

The global problem and its solution is intended to serve as an aid to medium and long range planning and it will be extended later by a scheduling of investments.

3. A two stage approach for the global problem

This paragraph containes a mathematical description of the global model, starting with the definition of each of its main components. Finally the solution method of the model is discussed.

3.1. Waste Watert Treatment Plants /WWTP/ For each of the existing and potential waste water treatment plants - the set of which is denoted by T - the following data are given:

- present capacity in m^3/day /0 if nonexistent/
- for each of the permitted sizes:

  investment cost
  fixed cost of treatment /depending on the size only/

- alternative technologies and corresponding costs.

3.2. Main Sewer Systems /MSS/. The set of potential and existing main sewers will be denoted by MS. Both gravity sewers and pressure pipes are included in MS, in our model they will differ from each other only in the finite number of permitted sizes and corresponding investment/operational costs. For the existing main sewers /the set of which is MS_E/ the smallest size is the present one with 0 investment cost. For each of the main sewers a quantity of own waste water is given obtainable from its districts by side sewers The quantity of own waste water of pressure pipes is zero, by definition. Naturally its capacity should by sufficient to carry on the inflowing quantity plus its own waste water, which adds up to the outflowing quantity, the inflow of the next main sewer.

A main sewer system /MSS/ is a subset of main sewers and pressure pipes, with the following properties:

/i/ It is a rooted and capacitated directed tree, i.e.: the elements of MSS consist of a directed, loopless connected graph with a single node having only incoming edg/es/ /called root/; there is a capacity associated to each of the elements.

/ii/ MS_E is part of MSS

/iii/ The capacity of each of the elements of MSS exceeds its own waste water + the /cumulated/ inflow;

/iv/ The root of the directed tree corresponds to a waste water treatment plant of sufficient potential capacity

A large number of MSS's are formed and the corresponding capacities, investment and operational costs are calculated. During the process of calculating the MSS's the ones, that are uniformly inferior than any of the previously generated ones, are automatically dropped. The remaining family of MSS's will be denoted by S.

It should be noted however, that each of the MSS requires a kind of engineering size calculation /including further details not mentioned here but done in the actual computer program/. A careful and interactive way of determining a good set S is an essential part of the efficient problem solving.

3.3. The global model

In the preparatory phase of the model a superfluous set of main sewer systems is determined as it is outlined in the foregoing paragraph. In order to ensure a higher degree of freedom most of the subregions are covered many times by the different main sewer systems. Additionally local and large regional MSS's should be included according to careful pre-studies of engineers and decision makers. In case of very large regions /e.g. the entire lake Balaton/ a set of MSS's containing a great number of sewer systems may be prohibitive from the point of view of computer time. In this case either less details may be considered or a kind of decomposition is advisable. Fortunately a natural decomposition into northern, southern and eastern shore can easily be done for the Lake Balaton. Beside studying larger regions, special problems of smaller regions can also be examined in substantially more details. The model described below can be applied to smaller and larger regions using different side constraints depending on the aim of the examination. In the model, the following notation is used:

Data of main sewer systems /MSS/

S	set of main sewer systems
x_i	0-1 variable, = 1 if MSS i is realized, 0 otherwise
c_i	invesment cost
o_i	operational cost /for 1 year/
f_i	waste water flow /m^3/day/
p_i	phosphorus load /kg/day/

Data of waste water treatment plants /WWTP/

T set of waste water treatment plants /existing and potential/

y_{jk} 0-1 variable, = 1 if WWTP j of size k is realized, 0 otherwise

d_{jk} investment cost

r_{jk} operational cost /size dependent, quantity independent part for one year/

K_{jk} capacity /m^3/day/

α_j percentage of phosphorus removal /parameter or variable/

$g(\alpha_j)$ operational cost of phosphorus removal

W_j set of MSS's ending at WWTP_j

U_j set of permitted sizes for WTP_j

Other data

N required number of sewered population units

q m^3/day equivalent of one population unit

S_v set of MSS's containing /sub/ village v

P permitted phosphorus load in the region

λ constant factor for comparing investment and yearly operational costs

V set of/sub/villages

The global model itself is formulated as a pure 0-1 integer programming problem:

$$\min \sum_{i\in S}(o_i + \lambda v_i)x_i + \sum_{j\in T}\sum_{k\in U_j}(d_{jk} + \lambda r_{jk})\, y_{jk} + \sum_{j\in T} g(\alpha_j) \sum_{i\in W_j} p_i x_i \qquad /3.1/$$

$$\sum_{i\in S} f_i x_i \geqq qN \qquad /3.2/$$

$$\sum_{i\in S_v} x_i \leqq 1 \qquad /\text{for all } v\in V/ \qquad /3.3/$$

$$\sum_{k\in U_j} y_{jk} \leqq 1 \qquad /\text{for all } j\in T/ \qquad /3.4/$$

$$\sum_{i\in W_j} f_i x_i \leqq \sum_{k\in U_j} K_{jk} y_{jk} \qquad /\text{for all } j\in T/ \qquad /3.5/$$

$$\sum_{j\in T} \alpha_j \sum_{i\in W_j} p_i x_i \leqq P \qquad /3.6/$$

$$x_i \in \{0,1\} \qquad y_{jk} \in \{0,1\} \qquad /\text{for all } i,j,k/ \qquad /3.7/$$

The objective function contains both the investment and the operational costs of MSS's and WWTP's, furthermore the /mainly chemical/ operational cost of phosphorus removal. The investment cost of tertiary treatment is included in the corresponding value d_{jk}, if the recipient water is the Lake. Constraint /3.2/ ensures the required level of sewerage. The next two inequalities exclude non-permitted duplications of main sewers and waste water treatment plants respectively.

Constraint /3.5/ is to avoid overloading of WWTP's, finally the inequality /3.6/ is a control of the phosphorus load of the lake. The latter can be substituted by constraints of the same type for certain subregions, if necessary.

4. A computer program system for handling and solving the global model

The program system was written for an IBM 3031 computer in APL language. All programs can be used both in interactive and in batch mode. For convenience, the interactive mode is supported by a conversational frame providing menus at each step. This question and answer system helps to choose the function of the system to be used next and to avoid data and parameter errors. Independently from the mode of operation, most of the data are obtained from a data base.

The main units of the program system can be divided into three parts /the modul name will be followed the short description of its main function/:

4.1. Data handling

UPDATE: input or modification of data /in the data base/

REGIONS: survey of data, output tableaux, maps, interrelations of data

RESULT: handling, analysis, condensed or detailed output tableaux and maps of solutions

4.2. Engineering calculations of main sewer systems.

ENGMSS : complete engineering calculations of a single MSS /sizes, costs, runoff time, total quantity of waste water and phosphorus etc./

GENMSS: generating and preselecting a large number of MSS's /control by input parameters or done interactively/

4.3. Forming and solving the global model for a given region.

PROBGEN: generating and arranging the data of a global model, on the basis of previously generated set of MSS's, given potential WWTP's and other planning parameters.

PROBSOL: the solution of the generated problem by a mixed integer algorithm /presently by MPSX, special algorithm is under development/

5. Applications of the global model

The global model can be used for various management and engineering purposes in both long and medium range planning of sewer systems, furthermore in analysing critical situations. A few of such situations are outlined below.

Strategic decisions in a 15-30 year long range planning process may be supported by an appropriate global model containing a large region. These questions may include alternatives such as

- local or regional sewer systems
- choice of recipient water /the alternatives may largely vary in operational and investment cost, effect on recipient water etc./
- building of multi-purpose reservoirs, etc.

Economico-engineering investigations may be carried on for smaller and larger region for example:

- cost/effectiveness studies of concurrent subsystems
- estimates of sewerage costs of developing areas
- adjustment of short and long term costs and goals.

The global model can also be considered as a tool in water quality management problems. In this case the prescribed water quality may be attained by coordinated actions of different fields /controlling of industrial, agricultural and recreational activities and their waste water/. The different actions on various fields have different costs and water quality improving effects. The result of a parametric analysis of the global model can be used in the overall comparison of cost/effectiveness among different actions on the mentioned fields. Another possibility is a large integrated model containing all aspects and components of the complex decision.

This can be partially substituted by an interactive approach using several models of various fields.

6. Preliminary results for the Lake Balaton resort area

First of all, it should be noted, that the examination of the entire catchment area would be very useful /especially that of the Zala river/. We have restricted our experimentation to the resort area because of the availability of its data and the better concentration of the research. All of our data /existing and potential location of main sewers and waste water treatment plants, investment and operational costs etc./ are based on the 1980 situation, some of them are estimated or approximated.

Fortunately, the Lake Balaton resort area can be decomposed into three almost independent parts southern, northern and eastern shore. Until now we have solved the global model 26 times for different regions and various requirements. In the preporatory stage several hundred main sewer systems were obtained /after preselection/. The first striking fact was the relatively low operational - mainly chemical - cost of phosphorus removal. For example a regional sewerage on the eastern coast would result in a 9.8 t/year phosphorus load of the lake. The chemical cost of the removal of the 75 %, 85 % and 95 % of this quantity would be 0,6, 0.8 and 1.2 million Ft/year respectively. The investment cost of phosphorus removal - in case of satisfactory secondary /biological/ treatment and in case of simultaneous phosphorus removal - consists of the dosing equipment only which is in the range of 2 million Ft. Even for medium size waste water treatment plants this should not be a considerably amount.

Interesting results were obtained in the question of local versus global /regional/ sewerage systems. The heuristic engineering idea, that the larger percentage of sewering is required the more preference of regional system is obtained, was supported by the series of solutions of southern shore. On the northern shore however the high cost of connecting the systems /the shore is dissected by mountains/ and the relatively high percentage of present sewering /only not very densly populated areas are unsewered/ resulted in many smaller waste water treatment plants and quite high per capita investment cost of sewering. For brevity's sake we shall compose only the northern and southern shore results. /Each of which is an aggregation of the solutions of 4-6 different problems, with various parameter values in the indicated range/:

		NORTHERN SHORE	SOUTHERN SHORE
number of existing	WWTP's	10	7
present capacity	m^3/day	28 200	16 430
present load	m^3/day	34 260	29 700
additional load to be treated	m^3/day	+ 9 000	+40 000

	NORTHERN SHORE	SOUTHERN SHORE
per capita cost of severing /thousand Ft/population unit /	15-20	7-11
number of WWTP's in optimal solutions	10-11	2-3
required percentage of sewered population /in the percentage of population that can be realisticly supplied by drinking water/	82%-86%	45%-70%

7. Conclusions

In the present paper a two stage approach for solving a model, which can be used as an aid for sewer systems planning, was described. The so called global model and the oomputer program system for handling and solving this model seems to be a versatile tool for management and engineering problems of medium and long range sewer system planning. It can also be used interactively, providing computer aid through many steps and iterations of the planning process. A further development for scheduling the investments - and compromising short and long term goals - seems to be both advantageous and feasible.

The experimentation of the system with the Lake Balaton resort area data shows, that it can also play an important role in the water quality management investigations of the lake /including cost/ effectiveness studies of phosphorus removal/ and in regional development studies. The future research should also include the examination of the catchment area of the lake.

Finally it should be mentioned, that most papers in the literature dealing with sewer systems concentrate on certain engineering subproblems like technology planning, determining the optimal size of water treatment plant, vertical alignement of pipes of predetermined location etc. In the present paper the emphasis is on global problem of location of elements with less details of optimal sizing, furthermore the vertical alignement problem is substituted by simple average numbers. The two approaches however can and should be combined in several ways, e.g. by pre-or post evaluation and preoise planning of some of the important components and comparison of near optimal solutions.

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FOOD CHAIN MODEL FOR SHALLOW IMPOUNDED RIVERS

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1. Introduction

Rivers have been channelized and impounded for numerous reasons such as barging, energy generation, flood water protection, etc. With the formation of a sequence of quasi-reservoirs along the river axis significant changes in water quality are to be expected.

Contrary to reservoirs with annual stratification these impounded rivers are shallow, have short detention times, and receive large amounts of partly treated water. In addition, however, they show highly eutrophic features.

Typical average water quality characteristics are as listed in Table 1 (taking as an example the river Main in the Fed. Rep. of Germany between 1979-1981 ($Q_{50} = 55\ m^3/s$)).

2. Modeling Purpose

Water quality modeling has become a valid instrument in planning and management in the area of pollution control. The study of such multi-purpose waterways should include a modeling exercise which supplies the response function of the system to (reduced) loads. Such loads originate either as primary

Table 1: Water Quality Characteristics of the River Main (FRG)

PARAMETER		AVERAGE	MINIMUM	MAXIMUM
Temperature		15.0	2.0	24.0
pH		7.8	7.0	9.0
$HCO_3^- + CO_2$	(mg C/l)	47.0	37.0	53.5
Susp. Solids	(mg/l)	27.0	6.0	62.0
BOD	(mg/l)	3.5	2.0	10.0
COD	(mg/l)	12.5	10.0	20.0
TOC	(mg/l)	3.8	2.5	6.0
DO	(mg/l)	9.5	4.5	20.0
Diss. P	(mg/l)	0.85	0.6	1.3
Total P	(mg/l)	1.0	0.7	1.3

load (e.g. in terms of BOD, COD, NH_4-N or phosphorus), or from primary-production, as so-called secondary-load (e.g. in terms of total carbon, oxygen or Chl-a).

The model proposed here provides a practical tool for management purposes taking into account a simplified biocoenosis with five components (Fig. 1) and the interactions between this 'living'-part of the model with the carbon, nitrogen, phosphorus and oxygen budget of the system. Furthermore, the influence of physical boundary conditions, such as temperature, light and hydraulics are incorporated.

3. Model Structure

Figure 2 shows a model-'detail', the substructure for the interrelation between phytoplankton and dissolved available

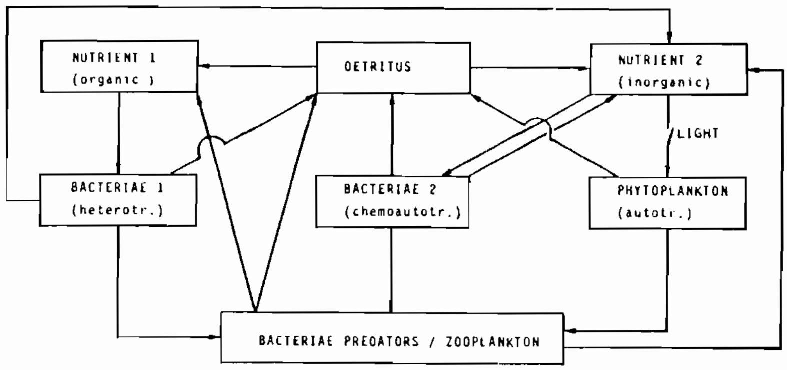

Fig. 1 Schematic structure of the model: interaction between biocoenosis and substrate.

phosphorus. Four state variables are used to describe the biochemical transformations. The mass flow rates are influenced either by the state variables or by environmental boundary conditions.

The reactor concept used for the computation determins the transport mechanisms and boundary conditions. Three types of reactors (shown in Fig. 3) are incorporated into this model.

The completely mixed reactor for example, can be used to 'calibrate' the transformation constants by simulating the observed data of controlled batch or chemostat experiments with river water. For the simulation of a real system with typical characteristics of one-dimensional flow (e.g. a 'chain' of shallow lakes), the second type of reactor will be used by incorporating the results of a proper hydraulic model for the description of the transport properties and the calibrated transformation rates from the simulations described above.
The third reactor type provides two layers, one layer accounts

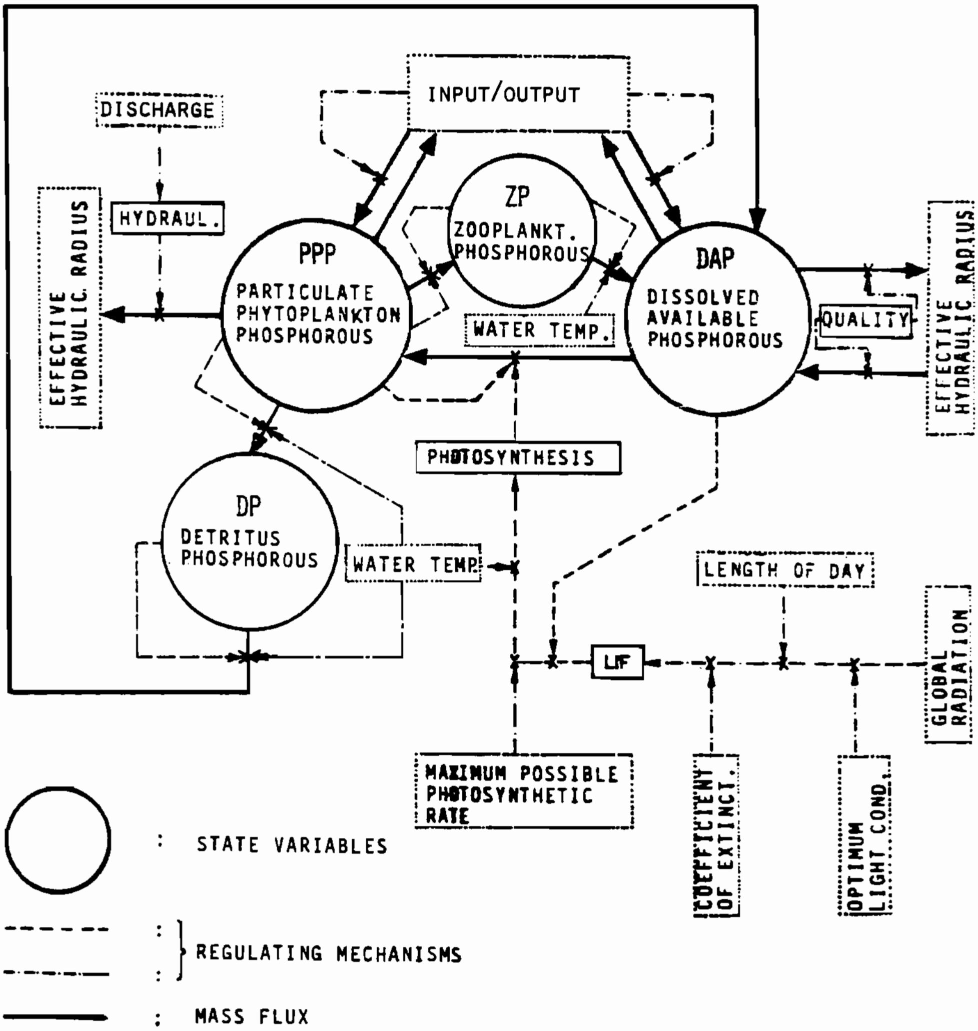

Fig. 2 Schematic structure of the model: matrix of relationships affecting phytoplankton-phosphorus.

for the main water-body as an aerobic reactor and a second one, a sediment layer, acts as 'active' zone or transition zone to the anaerobic bed. It can be used to model situations where a detailed consideration of the transformation and transport properties in the upper sediment layer cannot be omitted (e.g. dynamic of internal loading).

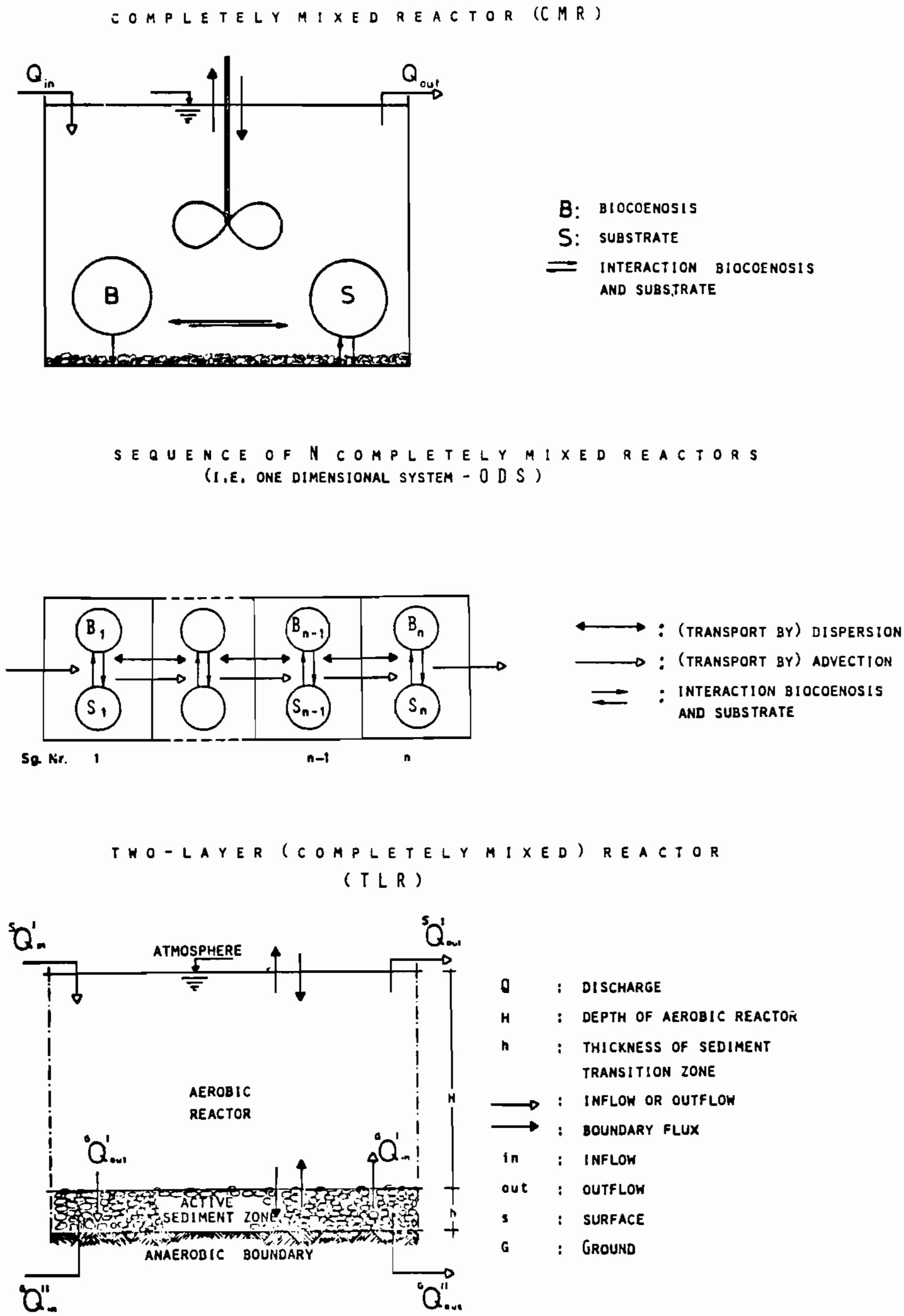

Fig. 3 Possibilities of model implementation.

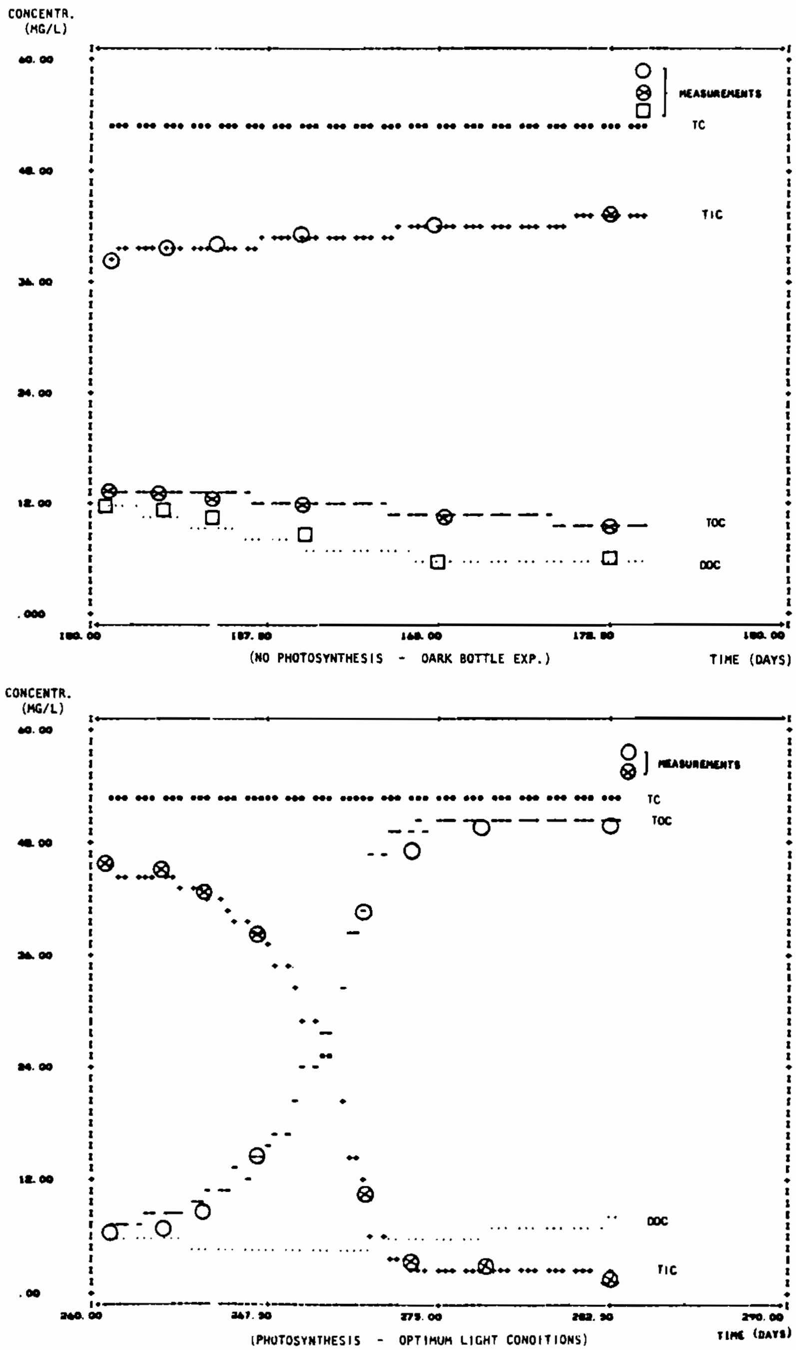

Fig. 4 Water quality changes described by model: inorganic and organic carbon (completely mixed reactor).

4. Model Application

The calibration procedure is based on a methodology of checking the model compartments separately against laboratory data (from long-term batch or chemostatic investigations for the transformation proceccces). Furthermore, data from in-situ studies (especially tracer investigations for the calibration of the transport part) are used and finally a synthesis is attempted to test the agreement of the whole model without changing parameters with river system data.

Figure 4 shows a plot of the calibration application concerning the carbon massbalance. The results of a long-term light-dark bottle experiment with river water (closed system) are used for this purpose. The dark bottles are used to calibrate heterotrophic and chemoautotrophic activities, whilst investigations under optimum light conditions permit to take primary production into account. A massive algae growth which occurs after the first eight days can be described indirectly by following the TOC and TIC changes. Similar plots concerning the nitrogen and phosphorus massbalances with additional description of the oxygen concentration may also be obtained.

5. Summary

Rivers have been channelized and impounded for numerous reasons. In comparison to reservoirs with annual stratification these impounded rivers are relatively shallow, showing highly eutrophic features. For such water bodies a model is proposed here which provides a practical tool for management purposes. It

takes into account a simplified biocoenosis and the interactions with the carbon, nitrogen, phosphorus and oxygen budget of the system. The shallow body of water is considered not to be stratified. The calibration procedure is based on literature data, own long-term laboratory investigations and in-situ studies. Preliminary results are also presented, mainly for discussion of trends.

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6. Literature

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ON THE UPPER LIMIT TO PHYTOPLANKTON PRODUCTION

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1. INTRODUCTION

Primary production, in particular phytoplankton production, is an essential source for the trophic web in aquatic ecosystems, representing therefore a topic of general interest in aquatic ecology (see Gessner 1960, Stein 1960a,b, Ryther 1963, Talling 1965, 1982, Bannister 1974a, Steel 1978, 1980, Uhlmann 1978, Smith 1980).

It is qualitatively well known that phytoplankton production is limited by the free enthalpy of solar radiation available for phytoplankton, i.e. by the so-called photosynthetically active radiation (PAR, 400...700nm) penetrating the water surface. All other environmental variables are here and in the following assumed to be in optimum. However, the numerical value of the limit to production resulting from the maximum daily sum of surface-penetrating PAR has been in the past often a matter of controversal opinions (compare e.g. the data given by Kok 1960, Uhlmann 1978, Steel 1978 and Talling 1982).

A closed semi-empirical solution to this problem has recently been derived theoretically by Baumert & Uhlmann (1983) using a combination of the approaches of Bannister (1974a) and Steel (1978). In the following we give a shortened explanation of our formulae and compare them with empirical production data, including those of lake Balaton.

2. THEORETICAL RESULTS

The nowadays field techniques applied to phytoplankton production measurements cover the whole range from those close to net data (oxygen bottles, grams O_2 evolved per day and square meter, see Vollenweider 1969) up to those close to gross production (^{14}C method, grams carbon fixed per day and square meter, see Gessner 1960, Dring & Jewson 1982). Hence, to derive an upper limit to measurement data from theoretical points of view, both net and gross production have to be considered.

2.1 Gross Production

According to Baumert & Uhlmann (1983) the approach of Bannister (1974a) to derive a theoretical upper limit to daily carbon gross fixation per unit area can slightly be generalized yielding

(1) $$\boxed{\Pi_g = (1 - \frac{z_{eu}}{z_*}) \cdot \bar{\Phi}_{max} \cdot \langle I_0 \rangle \Psi(p,q)}$$

where z_{eu} is the depth of the euphotic zone (m), defined by means of the vertical attenuation coefficient ϵ of PAR simply via the convention

(2) $$z_{eu} = 4.6/\epsilon$$

z_* is the upper limit to z_{eu} and given by (see Jerlov 1976 p.52, table XIII, Talling 1982 p.623, Baumert & Uhlmann 1983)

(3) $$z_* = \lim_{B \to 0} z_{eu} = \frac{4.6}{(1.2 \ldots 1.33)\epsilon_{w,min}} = 93 \ldots 104 \text{ m}$$

Here we assumed ϵ to be exclusively composed of a component due to pure water and of a component due to chlorophyll:

(4) $$\epsilon = \epsilon_w + k_a(B/\Theta)$$

k_a plays here the rôle of a specific extinktion coefficient of chlorophyll (see further Bannister 1974a), B is the biomass concentration (gC/m^3), Θ the carbon : chlorophyll ratio of the biomass and B/Θ consequently the chlorophyll concentration. The latter is, due to intensive vertical mixing, assumed to be uniformly distributed on the vertical axis.

With respect to (1) and (4) it has to be supplemented that the Lambert-Beer approximation was used to describe the underwater light climate. Therefore ϵ represents a combined daily, vertical and, in a certain sense, also a spectral average (compare Talling 1982).

Now the function $\Psi(p,q)$ occuring in (1) shall be explained.

It represents a dimensionless generalized form of the vertically and daily integrated photosynthetic rate:

$$(5) \qquad \Psi(p,q) = \frac{3}{2p} \int_{x=0}^{\pi/2} \int_{y=0}^{4.6\cdot q} \varphi\Big(J(x,y)\Big)\, dy\, dx$$

where

$$(6) \qquad \begin{cases} J(x,y) = p\cdot(\cos^3 x)\cdot e^{-y} \\ \quad p \;\; = I_{o,max}/K \\ \quad q \;\; = z_{mix}/z_{eu} \end{cases}$$

x,y are obviously auxiliary variables. The structure of $J(x,y)$ indicates the use of the Lambert-Beer presentation of the light climate (e^{-y}) as well as the application of a proposal of Straškraba (1974)about the diurnal course of PAR ($\cos^3 x$). $I_{o,max}$ denotes the diurnal maximum (instantaneous value) of the surface-penetrating PAR ($MJ/m^2\cdot d$), z_{mix} the mixing depth (m). $J(x,y)$ is obviously a dimensionless presentation of the vertical and diurnal course of underwater PAR. The function $\varphi(J)$ represents a dimensionless form of the commonly used photosynthesis-light relations where in our case also the light is dimensionless. Introducing many dimensionless variables instead of the immediate and clear physical variables seems to be an artificial approach. However the treatment of the formulae becomes easier and the final formulae become more compact and more transparent than using the original variables.

The function $\varphi(J)$ satisfies the following general conditions (for further details see Baumert & Uhlmann 1983):

$$(7) \qquad 0 \leq \varphi(J) \leq 1 \quad , \quad \varphi(0) = 0 \quad , \quad \left.\frac{\partial\varphi}{\partial J}\right|_{J=0} = 1$$

Particularly the last demand in (7) is an immediate consequence of the famous equation of Bannister (1974a)

$$\vartheta_{max} = k_a \cdot K \cdot \bar{\Phi}_{max}$$

which can be seen as a special local expression for the thermodynamic properties of phytoplankton photosynthesis where $\bar{\Phi}_{max}$ is, as also in (1), the ecologically effective maximum quantum yield of phytoplankton photosynthesis and ϑ_{max} the maximum photosynthetic rate (gC/d·g chl.). K plays the rôle of an initial slope parameter of the photosynthesis-light relations. (For a more comprehensive discussion with respect to different possible representations of $\varphi(J)$, particularly with respect to inhibition and saturation properties, see Baumert & Uhlmann, 1983).

To derive an upper limit to gross production we make use of the maximum possible photosynthesis-light relation $\varphi(J)$ which is given by

$$(8) \qquad \varphi(J) = \begin{cases} J \ , & 0 \leq J \leq 1 \\ 1 \ , & 1 \leq J < \infty \end{cases}$$

and evidently identically to the ansatz function used by Bannister (1979)

$$\varphi(J) = J/(1 + J^m)^{1/m}$$

if $m \longrightarrow \infty$. As a result of the general properties (7), $\Psi(p,q)$ can easily be shown to satisfy

$$0 \leq \Psi(p,q) \leq 1$$

and, particularly if $q \geq 1$,

$$\Psi(p,q) = \Psi_*(p) - e^{-4.6 \cdot q} \leq \Psi_*(p)$$

where

$$\Psi_*(p) = \Psi(p,\infty) = \frac{3}{2p} \cdot \int_{x=0}^{\pi/2} \int_{y=0}^{\infty} \varphi\big(J(x,y)\big) \ dy \ dx \leq 1$$

The particular choice (8) allows fortunately also the analytical treatment of the double integral, revealing

$$(9) \qquad \Psi_*(p) = \frac{3}{2p}\left\{\vartheta(1+\ln p) - 3 \cdot L(\vartheta) + p \cdot \left[\frac{2}{3} + \left(\frac{\sin^2\vartheta}{3} - 1\right) \cdot \sin\vartheta\right]\right\}$$

where

$$\vartheta = \arccos(p^{-1/3})$$

$$L(\vartheta) = -\int_0^{\vartheta} \ln(\cos t) \ dt = \text{Lobatshevsky function}$$

To derive an upper limit to (1) we have to solve the maximization problem

$$(10) \qquad \max\left\{\langle I_o \rangle \cdot \Psi_*(p)\right\}$$

where the daily integral $\langle I_o \rangle$ of surface-penetrating PAR (MJ/m^2d) is, as a consequence of the application of Straskraba's (1974) proposal, given by

$$\langle I_o \rangle = \frac{4}{3\pi} \cdot f_p \cdot I_{o,max}.$$

f_p is the light fraction of the astronomical daylength ($1-f_p$=dark fraction).Considering the meaning of p in (6) and (10), our maximization problem reads

$$\text{(11)} \qquad \max\left\{ \langle I_o \rangle \cdot \Psi_*(\varkappa \cdot \langle I_o \rangle\ ) \right\}$$

where $\varkappa = 3\pi/4K$. Via (9) it can easily be shown that the product $\{..\}$ is monotone increasing with respect to increasing $\langle I_o \rangle$ and that Ψ_* decreases monotone with increasing $\varkappa$ for a fixed $\langle I_o \rangle$. Therefore the solution of (11) is simply

$$\varkappa = \varkappa_{min} = 2.36/K_{max}$$

$$\langle I_o \rangle = \langle I_o \rangle_{max}$$

The upper limit to (1) is finally

$$\boxed{\begin{array}{l} \Pi_g = (1 - \frac{z_{eu}}{z_*}) \cdot \Phi_{max} \cdot \langle I_o \rangle_{max} \cdot \Psi_*(p_*) \\ p_* = 2.36 \langle I_o \rangle_{max}/K_{max} \end{array}}$$

The highest value K reported in the available literature was found in Jassby & Platt (1976, compare Jørgensen 1979):

$$K_{max} = 10\ MJ/m^2 \cdot d$$

and the maximum daily integral of surface-penetrating PAR can be estimated to be (see Bannister 1974b, Jerlov 1976, Sakshaug 1977, Straskraba 1980)

$$\langle I_o \rangle_{max} = 13\ MJ/m^2 \cdot d$$

These data yield finally

$$p_* = 2.36(13/10) = 3.1$$

and via (9) follows

$$\Psi_*(p_*) = 0.8$$

2.2 Net Production

According to Kok (1960), growth or net production is the difference between gross production and carbon losses (due to e.g. light and dark respiration and excretion as well). A summarizing description of these losses can be done by an overall loss rate with the units gC/d·g chl. Denoting the daily and vertical average of this loss rate by $\langle r \rangle$, the net production can be given by

$$(12) \qquad \Pi_n = \Pi_g - \frac{B}{B}\cdot\langle r\rangle\cdot z_{mix} = (1 - \frac{z_{eu}}{z_*})\cdot(1 - \frac{q}{q_*})\cdot\bar{\Phi}_{max}\cdot\langle I_o\rangle_{max}\cdot\Psi_*(p_*)$$

where q_* is the upper limit to $q = z_{mix}/z_{eu}$ and therefore defined by the so-called "column compensation point"(Talling 1957) $\Pi_n = 0$, i.e. by

$$(13) \qquad q_* = \frac{k_a\cdot\bar{\Phi}_{max}\cdot\langle I_o\rangle_{max}\cdot\Psi_*}{4.6\ \langle r\rangle}$$

The r.h.e. of (12) exhibits two "biological" degrees of freedom:

(i) z_{eu} , reflecting the biomass concentration via (2). z_{eu} is usually a known quantity in the case of production measurements.

(ii) q_* , reflecting the specific respiration rate via (13), including photorespiration! q_* is nearly always a fairly unknown parameter.

According to the approach of Steel (1978), there exists an optimum biomass concentration for which net production (12) is maximized, i.e. for which

$$\frac{\partial\Pi_n}{\partial B} = 0 \qquad \text{or, respectively,} \qquad \frac{\partial\Pi_n}{\partial z_{eu}} = 0$$

The latter demand will be satisfied by

$$(14) \qquad z_{eu} = \left(\frac{z_*\cdot z_{mix}}{q_*}\right)^{1/2}$$

Considering this condition, i.e. inserting (14) into (12) yields

$$(15) \qquad \boxed{\Pi_n = (1 - \frac{z_{eu}}{z_*})^2\cdot\bar{\Phi}_{max}\cdot\langle I_o\rangle_{max}\cdot\Psi_*(p_*)}$$

Fortunately neither q_{*} nor z_{mix} occur in our final formula (14) for phytoplankton net production in natural waters of the globe, which advantageoue property is a direct consequence of using (14).

3. COMPARISON WITH EMPIRICAL DATA

For the above-mentioned comparison a discussion of the ecologically effective maximum quantum yield Φ_{max} reveals to be necessary. From text books on plant physiology (Kok 1960, Radmer & Kok 1977) it is well known that the maximum quantum yield of photosynthesis is about (see also Bannister 1974a)

$$\Phi_{max} = \frac{1 \text{ mole } O_2}{8 \text{ einst}(685nm)}$$

This value was found in the lab for optimized internal states of synchronized cultures and for monochromatic light of optimum wave - length. Furthermore Φ_{max} was found to be essentially independent on temperature and species, almost a real "natural constant" of photosynthesis. Therefore the introduction of Φ_{max} into ecological modelling by Bannister (1974a) labelled a remarkable advance.

In natural waters several factors act in such a way, that the ecologically effective maximum yield is smaller than Φ_{max}, e.g.:

- The average spectral yield is smaller than the yield at 685nm
- The spectral energy distribution of underwater solar PAR is not a uniform one
- Asynchrony in natural populations as well as circadian rhythms lead to a further depression of the ecologically effective maximum yield, where the latter represents a daily average
- The assimilation number is usually smaller than 1, i.e. the carbon yield is smaller than the O_2 yield, possibly due to nitrate reduction.

These factors have been discussed by Kok (1960), Bannister (1974a) and by Radmer & Kok (1977).

By a comparison of our theoretical results with a set of extremely high empirical production data of many,very different natural waters of the globe (and of independently working authors, too, for the data sources see Baumert & Uhlmann 1983; a first version of the data set used here was published earlier already by Uhlmann,1978), we found Φ_{max} to satisfy possibly the relation

$$3 \text{ gC/MJ} \leq \Phi_{max} \leq 3.6 \text{ gC/MJ}$$

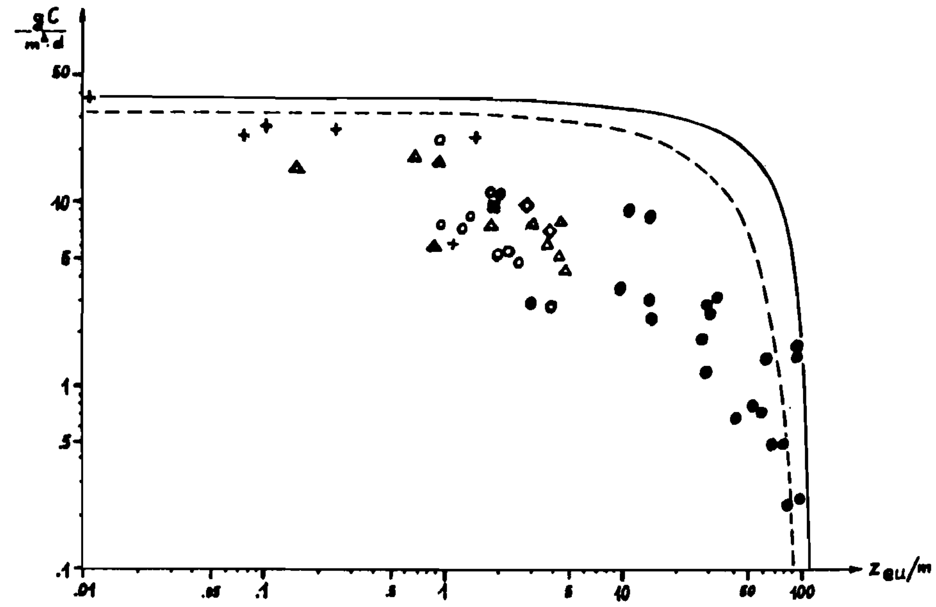

Fig. 1 Theoretical and empirical upper limits to phytoplankton gross and net production in natural waters, depending upon the euphotic zone depth.

---- $\Pi_n \leq (1 - \frac{z_{eu}}{93m})^2 \cdot 3\frac{gC}{MJ} \cdot 13\frac{MJ}{m^2 \cdot d} \cdot 0.8$

—— $\Pi_g \leq (1 - \frac{z_{eu}}{104m}) \cdot 3.6\frac{gC}{MJ} 13\frac{MJ}{m^2 \cdot d} \cdot 0.8$

+ Algal cultures, waste water treatment lagoons, artificial ponds

△ Freshwater and salt lakes

◇ Reservoirs, rivers and canals

o Coastal waters

● Other marine waters, in particular upwelling areas

■ Lake Balaton

In fig.1 the data set is presented graphically, together with our theoretically derived formulae for the upper limits to gross and net production. The fuzzyness with respect to z_*, to Φ_{max} and to the assignation of the data to Π_n or Π_g is vizualized by a range of uncertainty. The agreement is quite satisfactory and justifies the idealizations used. Particularly it demonstrates the on principle equal production potential of phytoplankton and terrestrial plant communities. On the other hand fig.1 shows the phytoplankton community of lake Balaton to be very productive (compare van Straten & Herodek 1982). In this respect it therefore can be compared with marine upwelling areas and other highly productive natural waters indicated in fig.1 .

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STATE-OF-THE-ART DISCUSSION ON EUTROPHICATION MODELING: SUMMARY AND CONCLUSIONS

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For the State-of-the-Art discussion questions and issues were prepared in advance. Subsequently a summary is given one by one on those subjects which were of general interest for the participants and where the discussion were completed with conclusions of general interest.

1. Are we in the position to develop "general models" which can be employed for various systems and objectives? If not, what are our present conclusions of general validity in the field of water quality modeling?

   It is neither practical nor desirable to develop general models. Data availability is one example of the many factors that make model generalization infeasible: the data requirements of a complex general model may not be tractable for all or even most applications. The exception to the infeasibility of generalization may be the generalization of particular model sub-parts for certain system properties.

2. Can we offer a general modeling procedure for handling the problem of a specific lake and its watershed?

   It is difficult to generalize even the methodology for model development or application since the scope of the effort will vary with the availability of manpower, time and funding for the modeling effort. Further, the objective of the modeling effort will influence the methodology. One recommendation which did emerge from the discussion is the universal requirement to relate the modeling effort with the data collection effort.

3. Incorporation of observations laboratory and in situ into the modeling framework. Data collection and feed-back with modeling.

   The most important comment to improve the connection between modeling and data collection was the proposal that field data collection for modeling purposes should concentrate on the rates of processes rather than simply measuring concentration. It was pointed out for example that although the concentration of orthophosphate P stays relatively constant at a low level, the rate of PO_4-P uptake or production may vary significantly. Along another vein, ecosystem enclosures were suggested as a potentially excellent method to collect field data under controlled conditions which allow the observations to be directly related to model formulation. Such experiments also allow to separate subprocesses i.e observations can be performed with-, and without sediment.

4. Role of uncertainties

   Uncertainty arises from input uncertainty, errors in the model structure and data uncertainty. Upon calibration these errors culminate in parameter uncertainty. When applying a model for predictions additional uncertainty arises from stochastic events mostly in the input forcing functions (meteorology).

   It is important to include uncertainties in the different stages of model development and application. In order to do this, however, an acceptable knowledge on the character and ranges of uncertainties is needed, otherwise just the degree of model freedom is increased.

   It was noted that the presence of uncertainty requires a management system which includes procedures for sequential action based upon feed-back from monitoring to adopt to more certainty gained as time proceeds. Management strategies should thus be investigative and mitigative to account for uncertainty.

5. Parameter estimation and identification. Sensitivity analyses

   It was felt that formal parameter estimation techniques were useful in some cases where sufficient data were available and where the models are sufficiently well structured and confined to tractable sub-processes. For large model systems automatic parameter estimation procedures may easily lead to a fit of the data to the wrong structure.Although theoretically the error sequence (i.e. the series of differences between model and data) should lead to diagnostic information. On how to improve the model structure (identification) little experience exists in the application of these methods in the field of eutrophication modeling.

6. Model validation and/or invalidation

The question of model validation provoked a vivid discussion. There was debate whether the common practice to calibrate an eutrophication model to observations of a particular year and test it for an independent data set for another year could be called validation. Rather, application to another lake should be the test. The majority however maintained thatthis was a valid procedure, because of the limitations in transferability of models to other lakes (see item 1). It was pointed out also that models were perhaps only valid for a narrow range of conditions, because of our fundamental inability to describe the complexity of the system completely in model terms. For example structural changes of ecosystems cannot be predicted on the basis of our present knowledge, and thus a particular model for a lake may remain valid only for a certain period of time.

7. The use of not yet validated models for management purposes: prediction of the response of the lake to changed nutrient loadings

For application for management decisions much depends upon the position on the cost-effectiveness function. For problems where we are on the flat side of the curve no severe requirements for validation have to be set. In eutrophication modeling at present a dynamic validation is hardly possible. However, all models, including the empirical Vollenweider approaches, show a similar response to P-load reductions. If it has been sufficiently demonstrated that phosphorus is indeed the limiting factor, no further precision is required in the majority of management problems, the more so since management alternatives are usually rather crude and consequently fine tuning is not realistic. In a different way, in the management context many - not necessarily "validated" - details of a complex eutrophication models are ruled out,and the solution of the management problem is often not too sensitive on (uncertain) parameters which were still preserved on this level of the analysis (e.g. internal load).

8. The relative importance of biochemical and hydrophysical processes in eutrophication. The proper description of these subprocesses. Time and spatial scales.

Time and space scales for biochemical and hydrophysical processes are often similar, and hence hydrophysics cannot be ignored. However, depending upon the desired degree of detail in space and time simplifications may well be possible if based on thorough investigation. The desired degree of detail should be related also to frequency and spatial distribution of loadings. There was some debate whether internal cell-quota models were more sensitive to rapid changes in loading or not, without a definite answer.

9. Wind induced sediment resuspension in shallow lakes

Although both daily field data obtained for Lake Balaton, and model exercises demonstrate that resuspension and subsequent de/adsorption processes do have a short-term effect upon algal growth in shallow, P-limited lakes, it was felt that long-term effects generally could be neglected.

10. Description of algal dynamics. Blue-green algae. Nutrient limitation. Experiments.

The fear that P-reduction would lead to more blue-green algae in Lake Balaton was found unvalid. Since the dominant species are N-fixing algae, their occurrance is likely to be caused by N-limitation, a situation that would cease to exist as soon as phosphorus is reduced. In general, however, little is known with respect to the occurrence of blue-greens.

11. The influence of (random) meteorologic factors. Their inclusion in modeling.

Meteorological factors can be introduced by including the long-term observed variances into the scenario generators, which generate a set of input conditions for the eutrophication model, as was done in the Balaton case-study. And although we cannot be sure about the chance that a particular event occurs (e.g. three consecutive "bad" years) we can specify the ranges of response of a model quite well. Perhaps more attention is needed for possible correlations in external factors affecting eutrophication (e.g. rainfall, runoff, nutrient load, temperature, solar radiation).

APPENDIX I: WORKSHOP AGENDA

Monday, 30 August

OPENING SESSION

Chairman: D.R.F. Harleman

Welcome - P. Benedek, as host on behalf of the Hungarian Academy of Sicneces and the National Water Authority

A. Hirsch, International Institute for Applied Systems Analysis

I. Kiss, Hungarian Committee for Applied Systems Analysis

MORNING SESSION: General Reports

Chairman: D.R.F. Harleman

L. Somlyódy: Major features of the Lake Balaton eutrophication problems. Approach to the analysis

G. Jolánkai and L. Dávid: Nutrient loads and watershed development

S. Herodek: Biochemical processes in Lake Balaton

Discussion

AFTERNOON SESSION: General Reports, cont.

Chairman: S. Orlovsky

L. Lijklema, P. Gelencsér and F. Szilágyi: Sediment and sediment-water interaction

D.R.F. Harleman and P. Shanahan: Hydrodynamics and mass transport aspects of Lake Balaton models

G. van Straten: Lake eutrophication models

L. Somlyódy: Lake eutrophication management models

Discussion

Tuesday, 31 August

MORNING SESSION

Chairman: L. Lijklema

T. Kutas and S. Herodek: BEM, a complex model for simulation Lake Balaton Ecosystem

J. Tóth and T. Kutas: A stochastic BEM model

M. Virtanen: Modeling of horizontal mass exchange in Lake Balaton

H. Baumert and P. Uhlmann: Upper limit of phytoplankton production

L.G. Tóth: Seasonal changes of the generation time of algal species in Lake Balaton

D. C. Lam and L. Somlyódy: An intercomparison between the Lake Erie and Lake Balaton water quality studies

Discussion

AFTERNOON SESSION

Chairman: S. Herodek

R.M. Baker and D.R.F. Harleman: Comparison of water quality models for Lake Balaton

P. Shanahan: Linked hydrodynamic - water quality model for Lake Balaton

G.L. Troubounis and H.H. Hahn: Food chain model for shallow impounded rivers

E. Matthaus and V. Wenzel: Usage of the interactive simulation system SONCHES for Balaton Modeling

H. Velner: Sedimentation and water quality of a shallow eutrophic lake

G. Schellenberger and V. Mohaupt: Reaction of a shallow eutrophic lake on annual input variations

N.A. Armand, V.F. Krapivin and F.A. Mkrthjan: Monitoring of water system and prediction their state by remote sensing and mathematical modeling

M. Straskraba: A dynamic management model using several anti-eutrophication measures

Discussion

Wednesday, 1 September

MORNING SESSION

Chairman: D.C. Lam

A. Urbányi and M. Bródy: Eutrophication and its effect on the water quality of the Fehérvárcsurgó reservoir, Hungary

L. Rákóczi: Correlation between lake currents and refilling of a dredged pit in the Keszthely bay

O. Győrke: Engineering interventions suggested on the basis of field observations and model tests to control silting and eutrophication

L.B. Kovács, E. Boros and F. Inotay: A two stage approach for large scale sewer systems design with application to the Lake Balaton resort area

Discussion

MORNING SESSION (contd.): State-of-the-art discussion on eutrophication modeling

Chairman: G. van Straten

AFTERNOON SESSION: State-of-the-art discussion on eutrophication modeling (contd.)

Chairman: P. Shanahan

Thursday, 2 September

MORNING SESSION: Meeting of scientists and decision makers

Chairman: P. Benedek

L. Somlyódy: Major results of the Lake Balaton Eutrophication Study

Discussion

AFTERNOON SESSION: Meeting of scientists and decision makers (contd.)

Chairman: L. Somlyódy

Discussion

CLOSING SESSION

Chairman: L. Somlyódy

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