International Institute for Applied Systems Analysis
IIASA www.iiasa.ac.at

Lake Balaton Eutrophication Study: Present Status and Future Program

Straten, G. van and Somlyody, L.

IIASA Working Paper

WP-80-187

December 1980

LAKE BALATON EUTROPHICATION STUDY: Present Status and Future Program

Gerrit van Straten
Lászlő Somlyődy

December 1980
WP-80-187

INTERNATIONAL INSTITUTE FOR APPLIED SYSTEMS ANALYSIS
A-2361 Laxenburg, Austria

THE AUTHORS

G. VAN STRATEN is with the Department of Chemical Technology, Twente University of Technology, Enschede, The Netherlands. Formerly, he was a research scholar and leader of the Lake Balaton Case Study at IIASA from April 1978-December 1979.

L. Somlyódy is on leave from the Research Center for Water Resources Development (VITUKI), Budapest, Hungary. He is a research scholar and the present leader of the Case Study at IIASA from February, 1980.

PREFACE

The deterioration of lakes' water quality due to artificial eutrophication is a typical symptom of our civilization. To stop this process and to improve the water quality in an economical way requires a systematic and coordinated analysis of many disciplines. The recognition of this fact related to Lake Balaton, one of the primary touristic resorts of Hungary and the largest shallow lake in Middle-Europe, initiated IIASA and the Hungarian Academy of Sciences in 1978 to establish a cooperative research on the development of ecological models and their practical application to the lake. In addition, it was felt that not only a contribution can be given to solve the problem of the Lake Balaton region but the methodologies developed can be generalized for other shallow lakes which represent, in general, a less studied field compared to deep lakes. Since it was also aimed to consider the water quality management problem (besides the understanding and description in lake processes), all the activities should be replaced to an opimization framework: one more aspect of general interest.

During the first half of the study many results were achieved. First of all, the establishment of a data base, the development of ecological models, and the increasing interaction among modeling, experimental work, and further data collection should be emphasized. Deriving from the nature of the problem, the modeling of the nutrient loading and water quality management is only at present in a progressing state and in the future, mainly, the activities of these fields should be strengthened.

The objective of the authors was to outline the present state of the cooperative research and define the activities planned for the future. This publication is closely related to the earlier background report CP-79-13 and jointly, they summarize all the important information related to the project. Also, the present paper can play a basic role in promoting and harmonizing the further research in the frame of the Case Study.

CONTENTS

LIST OF FIGURES AND TABLES

PART I. GENERAL OVERVIEW

1. Historical Background
2. Goals and Objectives
3. Organizational Aspects

LAKE BALATON EUTROPHICATION STUDY:
Present Status and Future Program

Gerrit van Straten and László Somlyódy

1. HISTORICAL BACKGROUND

The problem of eutrophication of lakes has received considerable attention in recent years. In most lakes phosphorus, and sometimes nitrogen compounds are in short supply for the development of algae and higher water plants. Some lakes are rich in these nutrient substances by nature. Such lakes are called eutrophic (i.e. nutrient rich). However, in the past decades lakes not previously eutrophic are showing a tendency towards increasing nutrient availability. Typical symptoms of this process of eutrophication are the sudden bloom of obnoxious algae species and the frequent occurrence of visible water coloration. Increasing discharges of domestic and industrial waste water, the heavier application of crop fertilizers, and the rise in airborne pollution are among the major causes of these undesirable phenomena.

The study of the eutrophication process and its causes has not only been prompted by serious concern about our natural environment, but also by direct economic interest. Floating waterplants and debris are a nuisance for the use of lake water for industrial or agricultural purposes, certain algae typical for eutrophic lakes, most notably blue-green algae, excrete toxic substances causing taste and odor problems in the production of drinking water; and, not the least, excessive algae blooms and overwhelming waterplants are hardly a pleasure for those seeking recreation in and on the lake.

In 1978 IIASA's Resources and Environment Area adopted the problem of eutrophication of lakes as one of its research topics. Many nations are facing the problem in a similar fashion, hence the exchange of experiences, methods and techniques beyond the national boundaries in an international setting, such as provided by IIASA, was expected to be 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 hydrodynamical processes involved, the strong interrelation between the various processes and phenomena, the stochastic variability due to meteorological influences, and the many side-effects of the possible alternative management strategies are just a few of the reasons why a systematic and comprehensive analysis is required. Another factor of interest is that lake system are usually so large, that in-field experimentation is very often impossible or subject to serious constraints. As a consequence, knowledge about the system has to be retrieved from whatever data are available at present. Data analysis, including mathematical modeling techniques is therefore an essential feature, the more so since data for this kind of systems are normally of rather poor quality. In addition, by virtue of the limited experimentation possibilities, our knowledge about the systems processes is far from complete, so that our models can be no more than very rough approximations of reality. In summary, the challenging questions the systems analyst is confronted with are: "How to model from inadequate data?", and, in advising managers: "How to manage from inadequate models?". In this respect the problem of lake eutrophication has interesting similarities to many problems in economics.

After a general workshop in September, 1977 (Beck 1978), IIASA's REN Area organized two consecutive technical workshops on the eutrophication of lakes. The first one, in December, 1977, dealt with deep lakes, the second one in April, 1978, considered the problems of shallow lakes (Jørgensen and Harleman 1978, and Jørgensen 1979). In moderate latitudes deep lakes are characterized by a prolonged period of a stable summer stratification, that is a warm top layer is floating on a cold bottom layer and mixing between the two layers is practically absent. Nutrients produced in the bottom layer by mineralization processes from sinking biological matter are trapped there, and therefore not available for algal growth in the light top layer, until in fall the stratification vanishes and the lake becomes fully mixed again. In the next season the cycle is repeated. This is why deep lakes usually show a remarkable year-to-year regularity in behavior.

In shallow lakes, the direct interaction with the lake sediments, enhanced by wind induced effects, introduces an additional dynamic dimension. The vertical mixing is normally very good so that nutrients swirled up might be available for algal growth immediately. Also the photic zone depth, i.e. the zone of algal productivity, is large compared to the total depth. Due to these factors shallow lakes exhibit generally more time variablility than deep lakes.

In the past many eutrophication models have been developed for deep lakes. However, the techniques of mathematical modeling and simulation are perhaps even more powerful tools in the study of the eutrophication problem of shallow lakes, where the dynamics are more complex as we have seen before. There is, moreover, considerable economic interests in such a study, especially in Europe where the majority of lakes in the non-mountaineous, densily populated areas are shallow. Thus, for REN's own in-house research it seemed appropriate to concentrate on a shallow lake study. Furthermore we felt that our methodological contribution would be most fruitful if applied to an actual case, and so the suggestion made by the Hungarian Academy of Sciences to adopt the Lake Balaton problem as a subject case study was readily accepted. The study started with a one-day task force meeting at IIASA in April, 1978, profiting from the presence of several of the participants of the foregoing shallow lake workshop, to discuss the plan for collaborative research envisaged for the case study.

2. GOALS AND OBJECTIVES

The study was officially started with the signing in May, 1978, of an agreement between IIASA and the Coordination Council for the Environmental Research on Lake Balaton of the Hungarian Academy of Sciences. It was agreed to "*establish cooperative links between their respective research groups to facilitate mutually beneficial activities aimed at the further development of ecological models and their practical application*". The actual implementation of this agreement was envisaged in the form of exchange of information, data and experience, the exchange of visits of scientists and researchers of both parties and the organization of scientific meetings at IIASA or in Hungary. In an attachment to the agreement it was established that the cooperation would be realized in the framework of 3 man-months visits of invited scientists to Hungary and 5 man-months of visits to IIASA.

The research proposal attached to the agreement summarized issues of the Balaton problem as follows:

-- Lake Balaton is a large shallow lake of 600 km^2 surface area, with an average depth of less than 3.5 m. The different areas along the axis of the lake exhibit various trophic states ranging from mesotrophic to eutrophic.

-- It is a phosphorus-controlled lake where both point sources as well as non-point sources play an important role.

-- The exchange of phosphorus between water and sediment plays an important but uncertain role in the eutrophication process.

-- It is an urgent task to propose efficient measures to control the eutrophication processes.

-- A reasonable amount of physical, chemical, biological, and hydrological data exists; however, a proper systems analytical utilization of the data is lacking.

On the basis of this list the areas of collaboration were identified as:

-- study of phytoplankton dynamics models and application and comparison of existing models. For this purpose the Balaton data would be made available for inclusion in IIASA's data bank;

-- activities on modeling the phosphorus exchange between water and sediment;

-- mutual exchange of information and cooperation in the field of nutrient loading modeling; and

-- elaboration of systems management models for eutrophication control.

A more detailed overview of the scientific work resulting from these objectives follows in Part II.

3. ORGANIZATIONAL ASPECTS

3.1. Collaborating Institutions

Instrumental in establishing contacts with IIASA has been the Computer and Automation Institute (SZTAKI) of the Hungarian Academy of Sciences (MTA), most notably the group of biomathematics, headed by J. Fischer, within the Department for Applied Mathematics. From the beginning essential direct links existed with the Biological Research Institute (BKI) of the Academy, situated on the Lake's shore in Tihany. Furthermore, an extensive network existed with individual scientists from a number of institutions in Hungary.

During the cooperation it soon appeared that the availability of data, especially with respect to the water quality and the loading to the lake, was crucial to a fruitful progress of the scientific work. Since large portions of these data were only available with the cooperation of the National Water Authority (OVH), and especially the Research Center for Water Resources Development (VITUKI) belonging to the Authority, there was a desire for a more official involvement of this institution in the research. Furthermore, VITUKI was in an excellent position to initiate additional field programs in the framework of the study. Thus, in January, 1979, a subcommittee of the Hungarian Committee for Applied Systems Analysis (MAREB) was founded in order to be responsible for the cooperation with IIASA with respect to the Balaton Case Study, in which the three institutions participate on a basis of equality. A member list of the subcommittee is given in Appendix A.

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.

3.2. Meetings and Scientific Exchange

In the framework of the agreement a total of 3 man-months visits by Hungarian scientists to IIASA was realized in 1978, whereas the total visits amounted to 7 man-months in 1979, indicating the acceleration of the project (a full list is presented in Appendix B).

During 1978-1979 three major meetings with outside participation were held, which proved to be very instrumental in discussing new ideas on the basis of results obtained, and to provide guidelines for further research.

As mentioned above, the case study began with a one-day task force meeting, April 17, 1978. Apart from 10 Hungarians and 10 IIASA scientists, 14 researchers from abroad attended this meeting. In general the research schedule laid down in the proposal attached to the Agreement was approved and valuable suggestions for actual work were given. The need for a rapid comprehensive collection of the data for inclusion in IIASA's data bank was particularly stressed.

A first joint IIASA-Hungarian Academy of Sciences Task Force Meeting was held in Tihany, Hungary, on the lake's shore, from July 26-28, 1978. Five participants from IIASA, three from abroad and about 15 from Hungary attended the three day meeting. Again the urgent priority of data collection and display was stressed. Several proposals for modeling and field work were discussed. With respect to organizational aspects of the project it was considered desirable to increase the proportion of biological, chemical, and hydrophysical field work participation relative to the mathematical involvement. This eventually led to the formation of the subcommittee mentioned in the previous section.

The second joint IIASA-Hungarian Academy of Sciences Task Force Meeting was held from August 27-30, 1979, again in Hungary, namely in Veszprêm. The total number of participants was 48, distributed among Hungary, IIASA, and abroad as 30, 7, and 11, around formal presentations, which were published in the Proceedings in 1980 (van Staten et al. 1980). The conference offered an excellent opportunity to report on results obtained during the existence of the case study, both with regard to experimental as well as to modeling work. Part of the status of this work is summarized in Part II of this report. The results were discussed in three working groups and the recommendations made are included in the Proceedings. Among others a plea was made for a better coordination of the hydrobiological and chemical measurement programs in the lake. Strong support was expressed for the continuation of the work on wind induced mass exchange and the sediment-water interaction, and additional suggestions were made to clarify the adsorptive capacity of the sediment. Finally, the continuing lack of reliable and comprehensive data on the lake's loading were felt as a serious hindrance towards better modeling, and it was strongly urged that this data be made available as soon as possible.

The experience, criticism, remarks, sugesstions, and recommendations obtained during the case study as indicated above, have been incorporated in the development of ideas on the study's continuation. These will be outlined briefly in Part III of this report.

PART II. SCIENTIFIC RESULTS

1. Introduction to the problem
2. Hierarchy of Models
3. Eutrophication Models
4. Nutrient Loading Models
5. Management Models

1. INTRODUCTION TO THE PROBLEM

Its nice summer climate, the fine scenery of the surrounding landscape, and its water quality have made Lake Balaton an attractive tourist resort of importance reaching far beyond the national Hungarian borders. With 600 km^2 surface area it is the largest lake in central Europe. In recent years scientists studying the lake have observed certain changes in the water quality of the lake. Clear signs of eutrophication became apparent and under unfavorable circumstances, algal blooms began to affect the touristic value of the lake, at least in the most polluted sections. This is, in a nutshell, the specification of the Lake Balaton eutrophication problem. This section summarizes the results of a problem analysis study, performed at IIASA in the framework of the Balaton Case Study. The aspects will be treated in a brief format here only, for details the reader is referred to the original publication (van Straten, et al. 1979).

1.1. General Characteristics

Figure 1 gives an impression of the Balaton and its catchment area. The lake is 77.9 km long and on the average 7.7 km wide. The surface area of the watershed is 5,180 km^2, that of the lake itself 596 km^2 (ratio 8.7:1). Its most typical feature is its shallowness: the average depth is 3.14 m, and the deepest point, the straights of Tihany, is not more than 11 m deep. Consequently, wind action is of great importance. Another consequence of the shallowness is the large temperature difference between summer and winter. On the average there is an ice cover for nearly two months in winter, whereas the temperature of the water rises rapidly to values around 24^0 in summer.

1.2. Hydrology

The principal river inflow is the Zala River in the southwestern end of the lake. The Zala drains about 2,750 km^2 or 53% of the total catchment area and is, therefore, of prime importance to the lake. The lake has only one single outflow, in Siófok, at the south-eastern end. The outflow is regulated by a gate, so that the level of the lake can be controlled within roughly 30 cm difference in most situations. Figure 2 gives

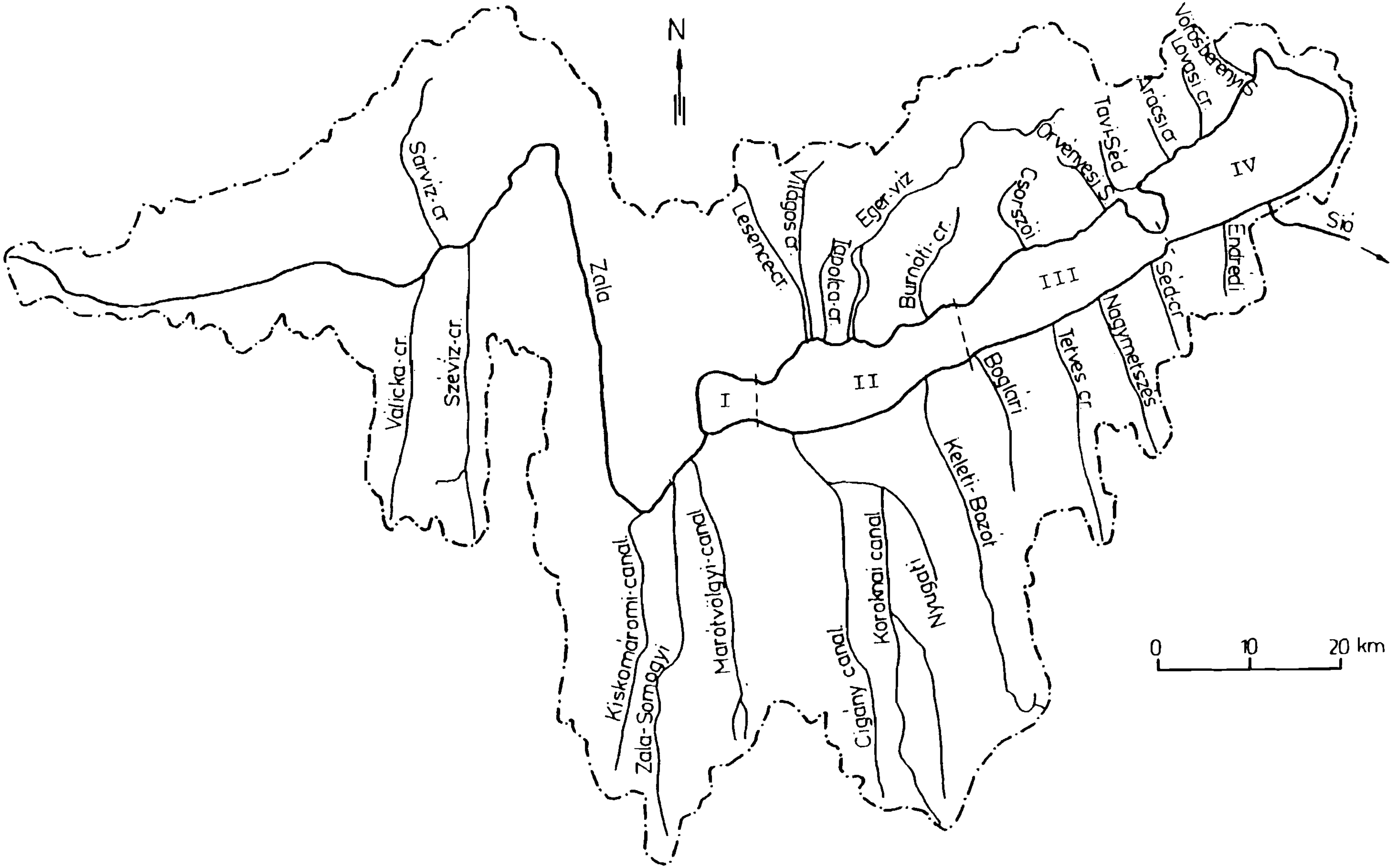


Figure 1. The catchment area and river system of Lake Balaton.

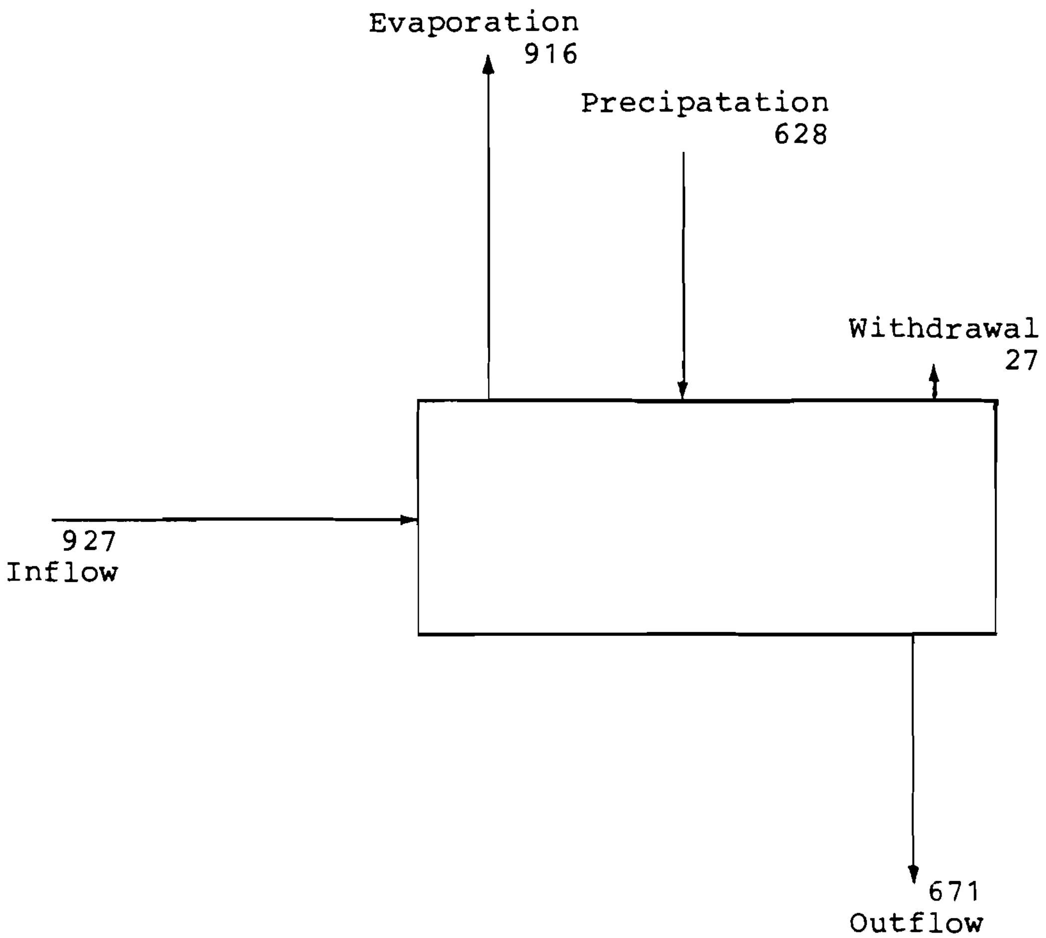


Figure 2. Annual waterbalance in mm.

the long-term yearly average water balance: the total inflow of 957 mm/year roughly balances the evaporation losses (916 mm/ year), whereas the outflow (671 mm/year) is about equal to the precipitation (628 mm/year). Given a mean depth of 3.14 m and a total annual input of 1.57 m, an average renewal time of 2 years results. However, this is just a lake-wide average. The renewal time is significantly shorter (1 year) in the Keszethely Bay (see Figure 1), and considerably longer (7 years) in the 'downstream' end of the lake (Siófok).

1.3. Hydrodynamics

Compared to the through flow derived from water balance considerations the wind-induced water motion plays a more important role. It is basically dynamic in character due to the strong fluctuation in wind (80-90 stormy days and approximately 1000 seiche events in a year). The spatial change is also essential and both the temporal and spatial variations are influenced by the surrounding mountains on the northern side of the lake. The prevailing direction is between NW and NW-N while the monthly average velocity ranges from 2 to 5 m/s. The windiest period is March and April, the calmest September (Hirling 1979).

Wind actions result in different kinds of free surface and water motions such as: set-up, causing a water level differences of as much as 1 m; seiches, with 5.5 and 1.0 h average period in longitudinal and transversal direction, resp.; and shallow waves, the height of which often exceeds 1 m (Muszkalay 1979). All these phenomena are influenced by the presence of the Tihany peninsula and can hardly be separated from each other.

Only limited knowledge is available on the velocity field. The motion at Tihany is determined mainly by unsteady velocities often higher than 1 m/s. In other areas the motion is less intensive. The experiments performed on a physical model (Györke 1975) indicated that for northern winds typical separated circulations were developed in different basins in the southwestern part of the lake. Seiches and set-ups may cause, however, a considerable mixing in the longitudinal direction among segments with different water quality.

with different water quality. At present no direct information about the extent of longitudinal mixing in quantitative terms is available.

Another important hydrophysical effect is the interaction at the bottom which allows dissolved and particulate materials to re-enter from the bottom layer. This and related phenomena will be discussed in Section 1.6.

1.4. Water Quality Characteristics

The dominant geological formations in the Balaton area are comprised of dolomite. The water is mostly of karstic origin and its composition is reflected in the chemical characteristics of the Balaton water. The calcium content is high: from 60 mg/l in the Keszthely end to 40 mg/l at the outlet. The magnesium concentration is of the same order of magnitude. In combination with the high m-alkalinity (4/5 mval/l) and the high pH (8.3-8.7) about 70% of the total calcium entering the lake precipates. These phenomena are closely related to the biological activity in the lake.

The differences in the calcium concentration along the lake show that a clear longitudinal gradient exists. It turns out that this is also the case for many other constituents. Figure 3 shows the yearly average and yearly maximum values at nine measurement points along the lake for chlorophyll, (as an indicator of algal biomass) and for some forms of the principal nutrient, phosphorus. Both total dissolved and total particulate phosphorus show a marked gradient, with concentrations decreasing from Keszthely towards Siófok. This is also mirrored in the chlorophyll data. Similar gradients are seen for suspended solids and secchi-disk depth. It is obvious that the Zala River plays a major role in this respect. On the basis of Figure 3 another important conclusion can be drawn, namely that wind mixing, although probably significant, is not strong enough to remove the longitudinal gradient entirely. Consequently, the lake can not be considered as fully mixed, which is important for modeling purposes. Thus far, a description with four segments has been frequently used (cf. Figure 1). It should be noted, however, that this selection was made on

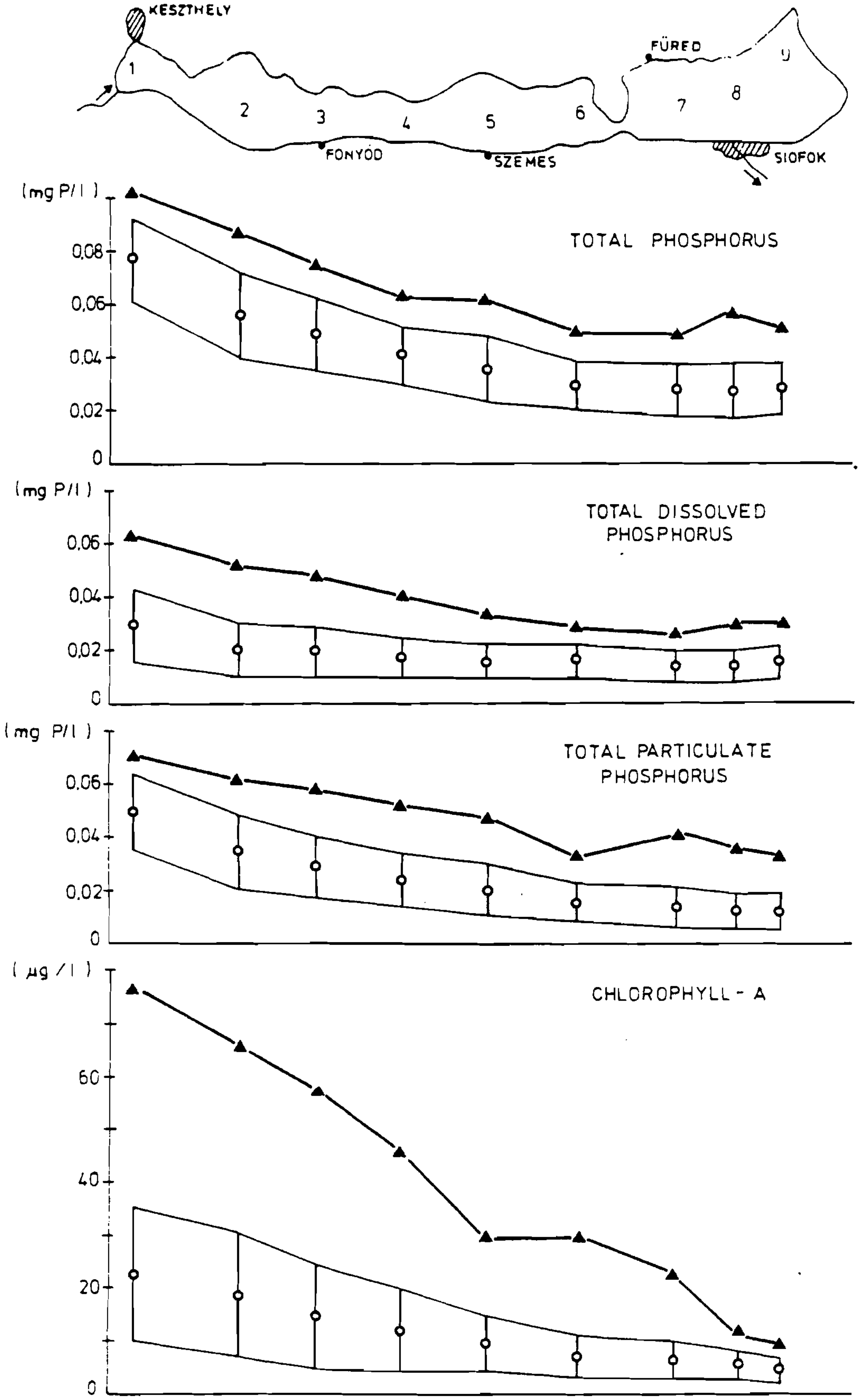


Figure 3. Longitudinal distribution of phosphorus and chlorophyll-a over the period 1976-1978.
o : average; bars: standard deviations;
▲ : maximum value

hydrological motives. It can not be excluded that on the basis of hydrodynamical or ecological considerations other segmentations may be preferred.

1.5. Phytoplankton and Phosphate Dynamics

It is interesting to have a closer look at the phosphorus and chlorophyll data. One significant aspect could already be read from Figure 3. This is the relatively high concentration of total dissolved phosphorus. Figure 4 shows the dynamics of the various phosphorus fractions, together with chlorophyll, for the years 1975-1978. The two lines for chlorophyll represent two different data sources. Again, the high proportion of the total dissolved phosphorus strikes the eye. It constitutes about 1/3 to 1/2 of the total phosphorus fraction. Since the ortho-phosphate is usually very low (2-5 mg/m^3) most of the total dissolved phosphate is of organic origin, or consists of condensed (poly)-phosphates. According to a recent publication (Dobolyi 1979) the latter can be a considerable fraction. However, very strong indications exist that both forms, whether dissolved organic or dissolved condensed phosphates, originate from decay of phytoplankton and other living organisms in the lake. Rapid demineralization by heterotrophic bacteria is likely to play a major role in the supply of ortho-phosphate needed to sustain the algal concentrations observed. Without this internal source algal growths of this extent would be impossible. Another possibility is that phosphate is released from the sediment. This will be discussed later.

From the time pattern in Figure 4 it is obvious that the lake's hydrobiology is highly dynamic, and strong differences both in pattern and in maximum algal concentrations exists from year to year. This is also confirmed by the available biomass data. Nevertheless, a general tendency is the occurrence of diatoms in spring, followed by a mixed phytoplankton dominated by Ceratium hirundinella in summer. In the Keszthely Bay and the adjacent Szigliget Bay blue-green algae (mainly Aphanizomenon flos-aquae) have started to be dominant in recent years. An impression on the development of the eutrophication process can be obtained from Figure 5, showing the primary production in the

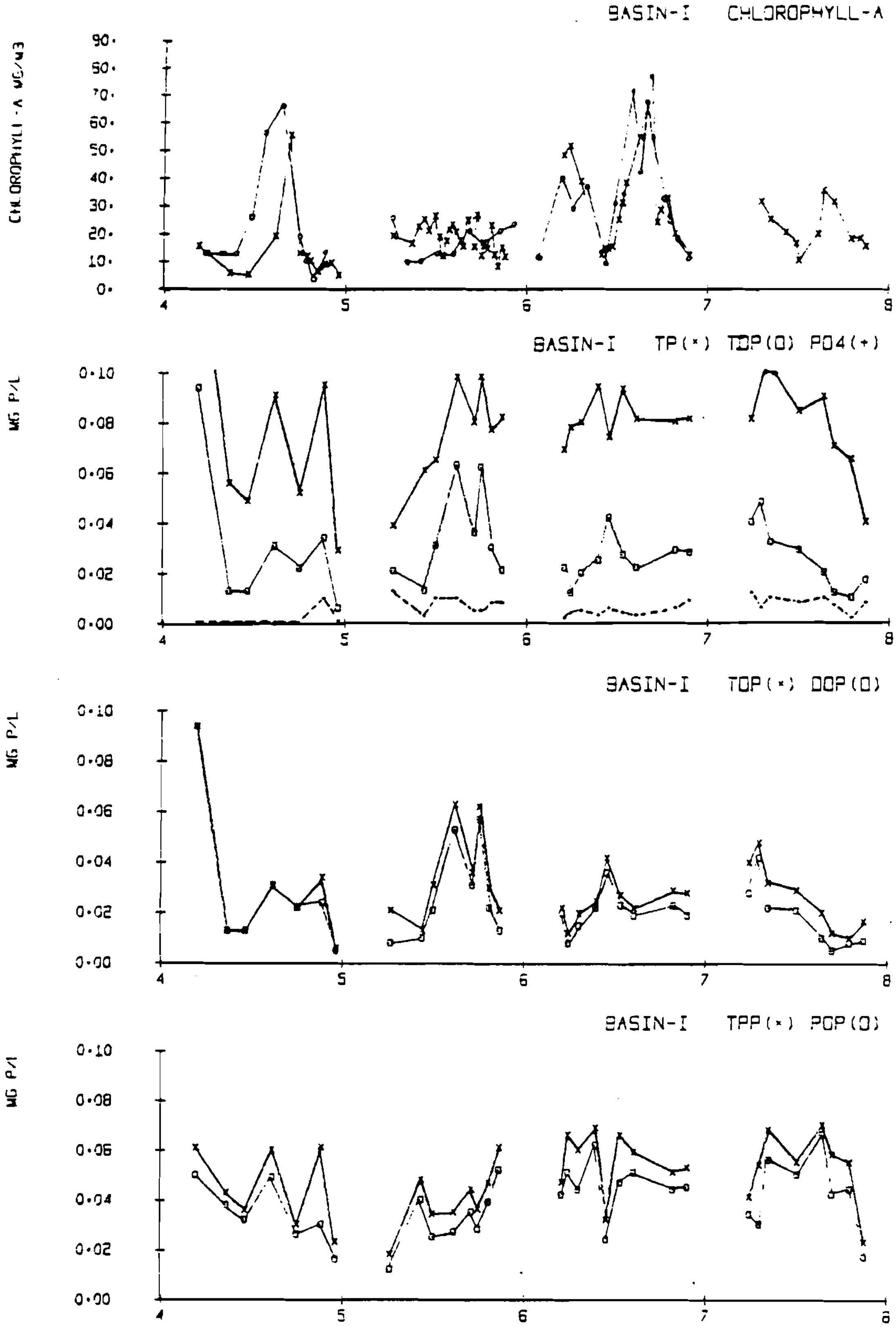


Figure 4. Chlorophyll and phosphate data for Keszthely Bay 1975-1978.

TP:	total phosphate	DOP:	dissolved organic/ condensed phosphate
TDP:	total dissolved phosphate	TPP:	total particulate phosphate
PO_4:	orthophosphate	POP:	particulate phosphate

Data for chlorophyll from two different sources (van Straten et al. 1979).

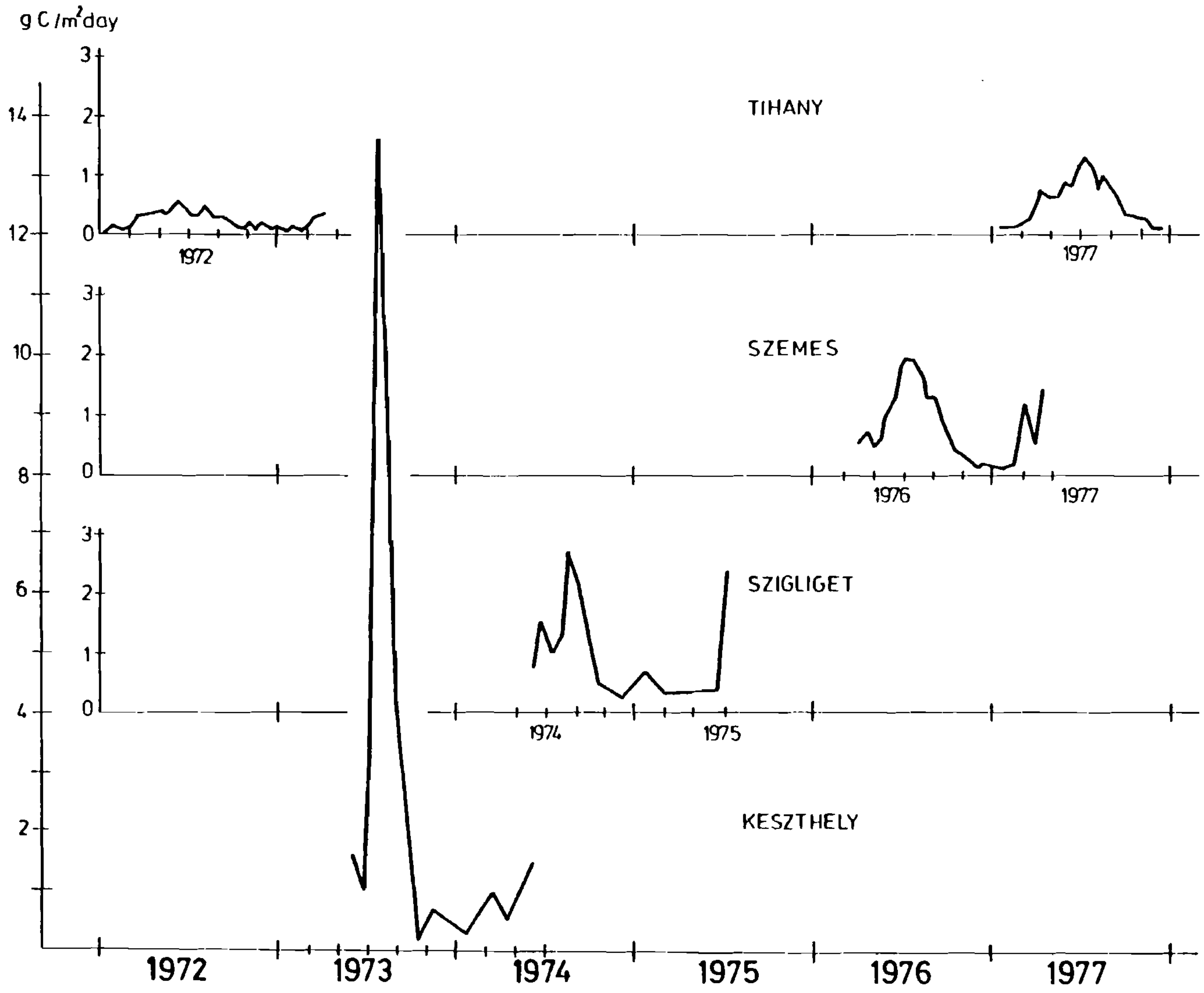


Figure 5. Primary production in the four basins since 1972.

various parts of the lake (cf. Herodek 1977). The most striking are the primary production values for Keszthely Bay in 1973, the peak value corresponding to 830 $gC/m^2/yr$, a hypertrophic value. The production to biomass ratio can be as high as 3-4 day^{-1}, which leads to theoretical maximum growth rates of 10 day^{-1} or more. These values are among the highest ever reported in the international literature. The actual growth rate is lower because of nutrient and light limitation. Due to the shallowness a steady interchange of suspended material with the sediment takes place under the influence of wind. The transparency is, therefore, highly fluctuating. Typical extinction coefficients are 2-4 m^{-1} (cf. Van Straten 1980). At the lake surface photoinhibition often occurs.

Given the high growth rates the question may rise: what processes are responsible for the death of algae? This question has not been fully resolved yet. A fact is that zooplankton concentrations are generally low, despite the favorable feeding conditions. The high suspended solid concentrations (in the form of calcium carbonate) is perhaps responsible for this.

1.6. Sediment and Sediment-water Interaction

A factor of considerable uncertainty in the study of the eutrophication process of Lake Balaton is the role of the sediment. Although most of the phosphorus entering the lake is stored in the sediment (80-90%) a very large proportion is probably not available for algal growth. Ecologically most significant is the top layer, that is the layer that could be swept away by wind induced currents and waves. It was calculated from oxygen measurements that roughly 1/3 of the total mineralization of dead organic matter takes place in the sediment. Even so, the sediment is relatively poor in organic material (only 2% of dry weight). The top layer is nearly always aerobic, except during short periods of microstratification in the Keszthely and Szigliget basin. Sediment analysis shows that a substantial part of the phosphorus in the sediment is bound to calcium carbonate, and also iron compounds seem to play a role (cf. Dobolyi 1980).

The role of the wind can be exemplified by looking at the water body data in March, 1977. In the Szemes basin measurements were conducted on two consecutive days, where on the second day a heavy storm prevailed. The data are presented in Table 1. Inspection of the different fractions clearly indicates that the observed increase in total P can be fully attributed to the particulate phosphorus fractions. Both particulate organic, as well as particulate inorganic phosphorus increased remarkably, whereas no significant increase in ortho-phosphate and other dissolved phosphate fractions was observed.

Table 1. Effect of a Storm on Water Quality Data (Szemes)

		28-3-1977	29-3-1977
Average wind speed	m/s	3	11
Suspended solids	mg/l	14	248
Chlorophyll	mg/m^3	19	37
Total phosphate	mg/m^3	28	160
Total dissolved phosphate	mg/m^3	11	10
Ortho-phosphate	mg/m^3	7	8
Total particulate phosphate	mg/m^3	17	150
Particulate inorganic phosphate	mg/m^3	5	65
Particulate organic phosphate	mg/m^3	12	85

The same results were obtained in a detailed study based on daily measurements (Somlyódy 1980). Thus, swirling up of sediment particles is a significant process. From the lack of increase of dissolved phosphorus fraction one would tend to conclude at first glance that the role of mixing of phosphorus-rich interstitial water is not very important. However, this need not nessarily be true. Certainly for the deeper parts of the lake, where interstitial dissolved phosphorus concentrations are typically in the order of 100 mgP/m^3, the observed erosion will lead to 10 more than 0.1 mgP/m^2 release, or for the whole lake 10 more than 5-10% of the total estimated dissolved phosphorus loading. But in the shallower near shore regions, much larger releases may be possible for two reasons: first, interstitial phosphorus concentrations along the shore are generally 5 to 10 times larger than at mid-lake points, and second, the wind induced erosion is large if the water depth is less. Moreover, a significant release of orth-phosphate may not be immediately discernable in the water body because of simultaneous and fast removal of algal uptake by adsorption on suspended solids.

It may be noted from Table 1 that a rise in chlorophyll was also recorded. This would be consistent with the observation that the sediment contains appreciable amounts of chlorophyll. For example, with the in-lake chlorophyll data no correction was made for the pheophytine content, and, consequently, the contribution of living algae in the resuspension term is uncertain. Theoretically, a rise in chlorophyll could also be due to increased longitudinal mixing during the storm event, because a sharp gradient from Keszthe-ley towards Szemes existed before the storm, but it is unlikely that this process could fully account for the doubling in concentrations.

1.7. Nutrient Loading

Because of the dominant role of phosphorus, increased loading of phosphorus in the past decades must be held responsible for the slowly deteriorating water quality. The population in the Balaton catchment area increased to roughly 420,000 inhabitants in 1975. Development in the region has lead to a larger degree of public water supply (56%), and although the proportion of the total population connected to a sewage system is still not very high (27%), it is likely that the development in this respect has contributed considerably to the nutrient problem of the lake. In addition, improved methods in agriculture, the major activity in the watershed, has entailed the use of more fertilizer, amounting to 70,000 tons

P_2O_5 equivalents in 1975. Other agricultural activities such as intensive live-stock breeding are also significant. Finally, in recent years, tourism around the lake has grown tremendously, and on peak days during summer more than half a million people take part in recreational activities in the direct vicinity of the lake's shores.

Direct data on the nutrient loadings are relatively scarce. The regular monitoring program contains rather detailed data for the Zala River (cf. Joó 1980), and from these data extrapolations to other sub-watersheds can be made based on surface area and slope conditions. In this way the ratio of the non-point sources over the four basins was calculated as 1:1:0.45:0.3 from Keszthely to Siófok. For sewage loadings the estimates for the lake total had to be interpolated over the four basins with the population density and sewage system connection ratio as the main guideline. Apart from the ortho-phosphate loading from the Zala River the distribution was estimated as 0.1:0.25:0.25: 0.4. The ultimate distribution of the loadings, such as used in segment oriented water quality models (BALSECT and SIMBAL, see Section 3.3.) is summarized in Figure 6.

The role of the various sources as well as the estimate for the present (base 1975) total and available phosphorus load is presented in Table 2. It should be stressed that these data are subject to relatively high uncertainties (cf. also Jolánkai, 1980). This is even more true for the column on 'available' phosphate. The annual total phosphorus load to the lake is estimated to range between 700 and 1,600 kg P/day, of which roughly 430-860 kg P/day is likely to be readily available for algal growth (ortho-phosphate). The total loading estimates are equivalent to a yearly surface loading of 0.4-1.0 g $P/m^2/yr$ in total phosphorus. However, about 1/3 of the loading is received by the Keszthely Bay, having only 6.4% of the total surface area. Consequently, the surface loading of total phosphate for this part of the lake amounts to 2.2-4.9 g $P/m^2/yr$, a very high value.

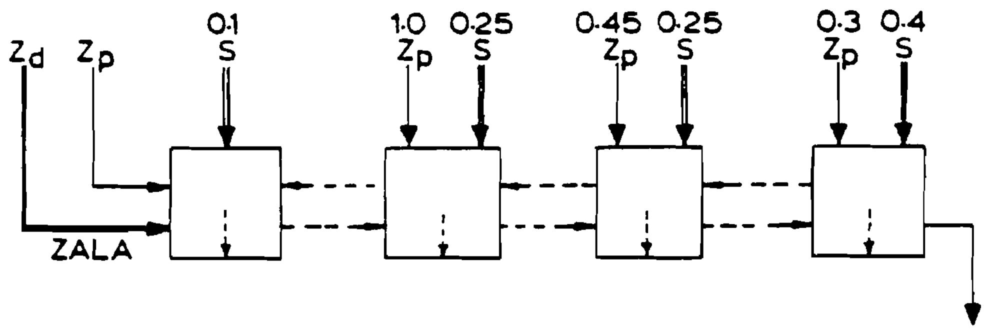


Figure 6. Distribution of P-loading over the four Balaton segments

S = estimated sewage load for whole lake (time constant, but higher during tourist season);

Z_d = measured ortho-phosphate load by Zala River (time varying)

Z_p = measured particulate phosphorus load by Zala River (time varying)

Table 2. Tentative Estimates of Present Total and Available Phosphorus Loading of Lake Balaton

	Total-P		"Readily Available"-P		
	kg/day	% of total*	Fraction readily available	kg/day	% of total*
Fertilizer loss, erosion, run-off resuspension	250-800	46	0.2	50-160	16
Liquid manure	50-200	11	1.0	50-200	20
Industry	30-50	4	1.0	30-50	6
Direct sewage	150-200	15	1.0	150-200	27
Indirect sewage	80-160	10	0.9	70-140	16
Precipitation	140-190	14	0.6	80-110	15
Total	700-1600			430-860	

*Percentages based on median values.

2. HIERARCHY OF MODELS

The introduction of the previous chapter demonstrates the complexity of the water quality problem. A successful design of models as an instrument for management purposes would demand a balanced integration of each of the relevant aspects, and requires a framework to place each of the individual elements of the problem into the perspective of the whole. Such a framework is presented in Figure 7. The basic idea is to first decompose the complex structure in smaller, more tractable units, accessible for separate and detailed study. A decomposition that directly comes in mind is the distinctions between lake and watershed. Indeed, the water quality problem itself lies in the lake, but the causes, and practically all control possibilities, are found in the watershed. In the next steps, the various studies on the basic level have to be put together on a successively higher level of integration. In these steps it is likely that only the essentials

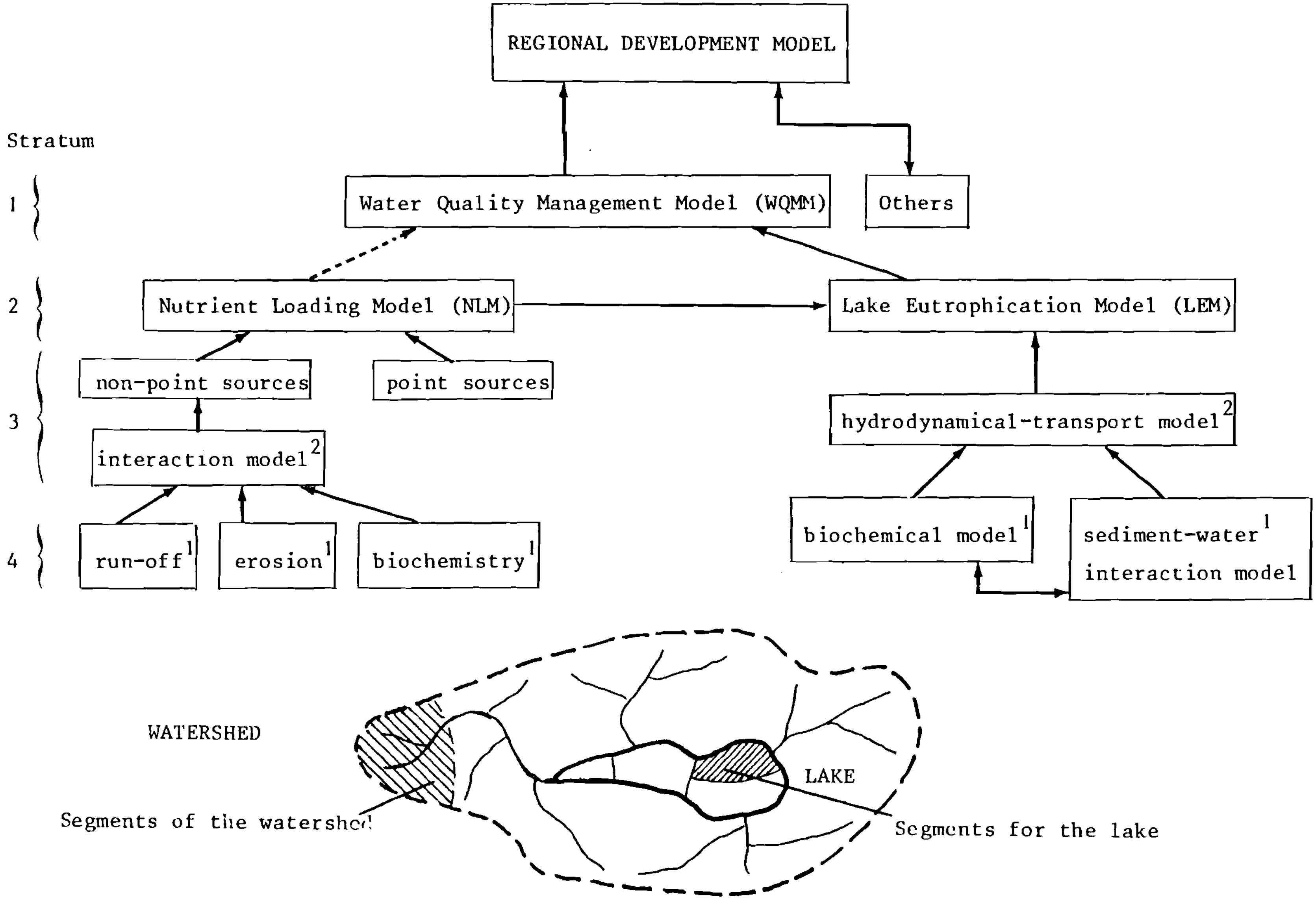


Figure 7. Hierarchy of models
1) submodels for uniform segments and 2) coupling of the submodels.

are preserved and that part of the unnecessary detail is ruled out. How this should be done properly is a subject of considerable scientific sophistication, in which present intuition and feeling for proportions still plays a significant part.

Returning to Figure 7 let us discuss the ideas outlined above in somewhat more detail. The fundaments of the analysis consist of the detailed studies of chemical, biological, and physical processes at the lowest level of integration. Even on this level simplificatins of reality are sought by employing modeling techniques, so as to ensure that the most significant features of the system behavior are captured by the models. It is obvious that these models rely heavily on field observations and specific field- and laboratory studies. Generally one would find try to isolate parts of the system for this purpose to exclude disturbances by geographic differences (e.g. soil types, slope conditions, load use, pollution loading, wind exposure, etc.). According to this line each of the models marked "1" in Figure 7 has to be developed for more or less uniform watershed and lake segments. It should be noted, however, that these are no hard scientific methods to decide how uniform 'uniform' should be, and experience and data based judgement are still important here.

On the next level (marked "2") the segment-oriented submodels are linked together. For the watershed this is the interaction and transport model which combines the sectional run-off and erosion including the carried and tranformed pollutants into the non-point sources (tributary sources) to the lake. In the lake itself the sectional biochemical and sediment models are coupled by the hydrodynamic transport model.

In the next level of integration the non-point sources are combined with the point sources into the nutrient loading model, which provides the input for the lake eutrophication model based on the lower level lake submodels. On this level simulations can be made of lake water quality under various alternative management scenarios, thus providing the information to be used in a water quality management model for the of the best, the optimal or non-inferior solutions (see below). Finally, this water quality management model could be thought of being a part of a regional development policy model, in which water quality is considered as but one element out of the total spectrum of aspects influencing regional development. Work on this type of model, however, goes beyond the competence of the Resources and Environment Area and will not be discussed in this report.

Figure 7 structures the various modeling activities in various strata, finally leading to the integrated water quality management model at the tip of the hierarchy pyramid (first stratum). What this representation does not show is how the management model feeds back to the lower strata when it comes to real application in decision making. Typically, the purpose of the decision model would be to generate alternative management options and strategies, and to select among these alternatives on the basis of one or more objectives. The feedback in this process of management alternative generation and selection may be illustrated by the following simple example. This example will also serve to illustrate the application of the lower strata of the decision hierarchy: the nutrient loading and lake eutrophication models. In presenting the example, we refer to Figure 8 which shows the elementary structure of Figure 7 in a simplified fashion for four alternative management structures.

Part (a) of Figure 8 indicates that every management alternative (generated by the management alternative generator, MAG) leads to a certain nutrient input (inp) to the lake and a resulting water quality (wq). In this example, we assume that the effects of the management strategy upon the input and upon water quality may be computed with the help of the nutrient loading model (NLM) and the lake eutrophication model (LEM), respectively. Hence, these lower strata models occupy a place of importance in the decision structure hierarchy.

With less predictive ability of the nutrient loading and eutrophication models of the second stratum, and the associated lower-level submodel one might attempt to bypass these usually complex elements by employing a direct relationship between management and water quality (dashed line in Figure 7), such an attempt has, in fact, been made for the Lake Balaton Case Study and is described more fully in Section 5.1.

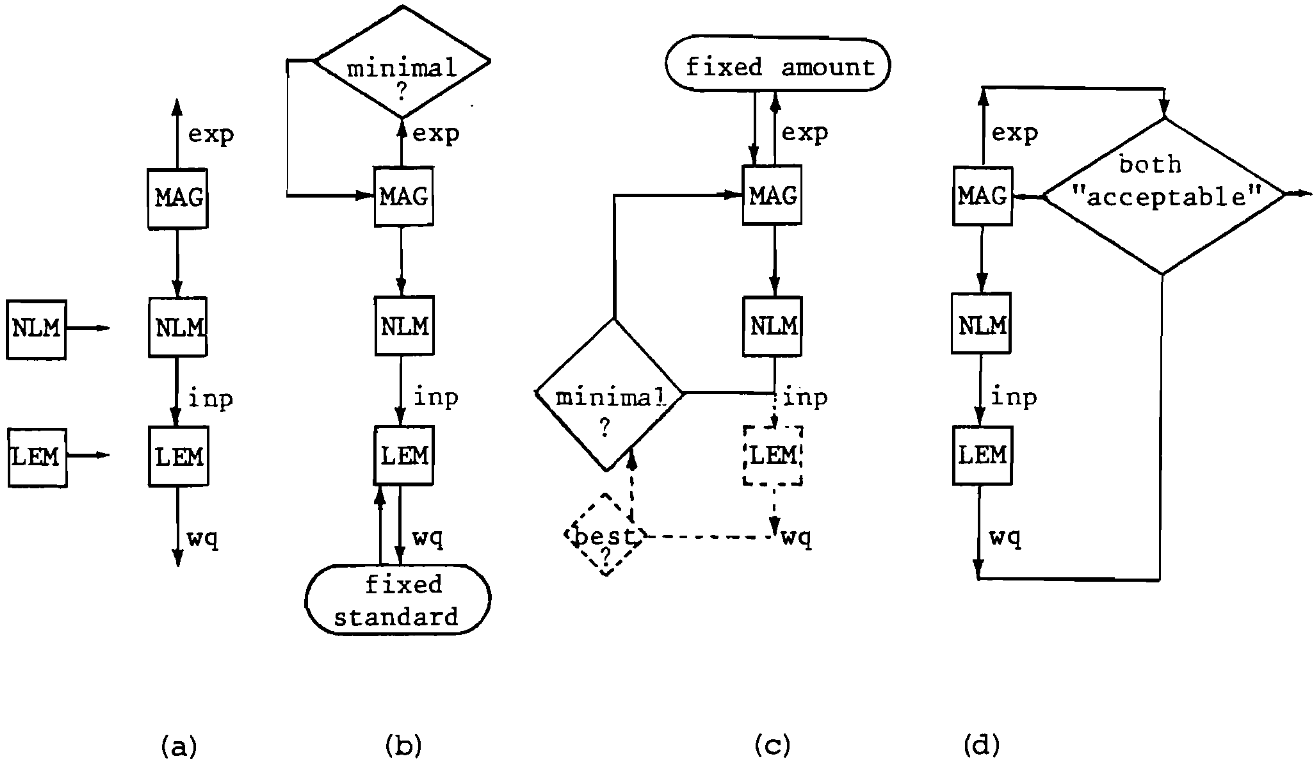


Figure 8. Various simplified management decision structures.

MAG = Management Alternative Generator

NLM = Nutrient Loading Model

LEM = Lake Eutrophication Model

inp = lake nutrient input

wq = lake water quality

exp = total expenses (including imponderabilia)

(a), (b), (c), and (d) — see text.

Each possible management alternative is, of course, associated with expenses (exp), and these are also indicated in Figure 8. We use the term expenses here, rather than costs to emphasize that the important consequences of a management decision may include more than just capital and operating costs. For example, agricultural loss, loss of natural preservation area, social impacts, and other so-called *imponderabilia* must be considered.

The parts (b), (c), and (d) of Figure 8 are further examples to demonstrate the various ways in which the basic structure can be operated. For instance, one might desire to set a specific water quality standard (e.g. algal biomass), which then serves as a constraint condition (Figure 8b). One then selects a solution that minimizes the expenses under this constraint. In Figure 8c a fixed, prescribed amount of expenses is assumed. Here the solution will be selected which minimizes either the inputs or the water quality variable. Note that in this case the lake eutrophication model merely serves as a check on whether the obtained minimum input, given the allowable expenses, leads to an acceptable improvement of water quality or not. In the examples (b) and (c) only one single criterion was used to select an optimal management strategy. In many practical situations this is not fully realistic, and a multi-criteria problem of the type of example (d) must be solved. Here some weighting of water quality improvement and required expenses is necessary. It should be noted that the management alternative generator must be designed in such a way that solutions for which both criteria can be improved simulataneously (so-called inferior solutions) are rejected right away. As a final remark note that the examples given here are of the utmost simplicity. In fact, even in the cases (b) and (c) some kind of procedure as given in (d) is unavoidable if the *imponderabilia* are to be accounted for in a proper way.

3. EUTROPHICATION MODELS

3.1. Hydrodynamics

The effect of wind induced water motion on lake ecology may be classified into two categories (cf. Somlyódy 1979). The first one is the entrainment of different substances from the bottom layer (see Chapter 1) including sediment and pore water. A brief discussion of modeling attempts on the interaction problem will follow in the next section. The second effect relates to the transport of dissolved and particulate materials in the water. This process is governed by the wind-induced circulation and determines the mixing of nutrient or pollutant rich inflowing waters with lake water and the mass exchange in a global sense among different lake segments. This second process is of great interest from an ecological point of view. To estimate its role, the analysis of time scales is a useful tool. Harleman and Shanahan (1980) found that diffusion and circulation has approximately the same time scale as biological and chemical transformations between a day and a week while hydrological through flow operates on a much longer scale. Similar conclusions may be drawn from other, simple calculations. When assuming, for example, typical wind-induced velocity profile with backflow in the vertical, the renewal time can be estimated for the Keszthely Bay as 5-10 days (W = 5-10 m/s). On the basis of Muszkalay's measurements (1979) a set-up with duration 12 h and W = 10 m/s will convect approximately 10% of the basin's total water voluem to its neighbor basin. After the wind decreases, the convection is followed by an oscillation causing dispersive-type transport. All those factors underline the significance of wind-induced horizontal transport, although on the average this transport is not sufficient to level out the gradients along the axis of the lake, as shown in Figure 3.

Principally two different possibilities exist to describe mass transport processes in a lake: (i) to perform detailed measurements in space for some conservative substances or, (ii) to develop mathematical models. In the absence of appropriate data here the second approach will be reviewed.

Mass transport is basically determined by water motion, consequently the computation of the velocity field must be done first on the basis of the momentum and continuity equations. Then, this can be followed by the application of the advection-diffusion equation. The numerical representation of the first part is called the hydrodynamical model, while the second is called the transport model, though in practice, they are applied simultaneously. In the subsequent paragraphs, hydrodynamical models will be discussed since their development is much more problematic.

The basic difficulty is caused by the need to solve a set of nonlinear partial differential equations (which may also include equations for deriving the eddy viscosity: this is the so-called closure problem) with generally complicated boundaries and boundary conditions. This solution needs considerable computer memory and execution time. Accordingly, a detailed coupling with biological models is not realistic. A simplification of the problem may be realized by eliminating dimensions along which velocity changes are non-essential. This procedure requires, however, detailed velocity field measurements which are not available here. Thus, one must start with a three-dimensional approach, calibrate it to water level and existing limited velocity data, and then to try and simplify the model depending on the lake behavior.

The literature includes a wide range of models, which were thoroughly reviewed by Watanabe et al. (1980). Considering three-dimensional versions only, the models may be classified according to the following criteria (multilayer models are excluded here): time dependency (steady or unsteady), simplification of the governing equations (e.g. neglection of nonlinear and horizontal diffusion terms), solution technique (mixed analytical-numerical or numerical methods such as that of the explicit or implicit finite differences and finite elements), boundary conditions (slip or non-slip conditions, rigid lid approximation, etc.), and treatment of turbulent eddy viscosity (Boussinesq assumption with empirical parameters or more sophisticated turbulence submodel). In most cases, the solution technique is closely related to the physical assumptions made; this fact explains the large variety of existing models.

In the frame of the Case Study three approaches were tested. At the very beginning an effort was made to apply the steady state, mixed analytical-numerical model of Gedney (1971), the computer program of which originally was designed for Lake Erie (Durham and Butler 1976). However, the structure of the program is closely connected to Lake Erie, causing the model adaption to Lake Balaton's geometry to be too tedious and preventing actual results from being obtained.

Another, more general model developed earlier for vertically stratified cooling ponds was brought to IIASA by Dr. V.I. Kvon (Institute of Hydrodynamics, Siberian Branch of the USSR Academy of Sciences, Novosibirsk, USSR). The model includes the non-linear convective terms, applies the hydrostatic approximation for pressure distribution along the vertical and ignores only the horizontal turbulent diffusivity. A finite difference solution technique is used. The closure problem is solved by using a two-equation $k-\varepsilon$ turbulence model (Launder and Spalding 1972, Watanabe et al. 1980), which involves two additional equations for the turbulent kinetic energy and its dissipation rate, respectively.

In practice application to Lake Balaton the maximal number of grid points is limited by the IIASA computer (221 horizontally and 5 vertically). Furthermore, the program may employ a stepfunction wind input history only, which limits the application to the simulation of set-up events. Results indicate that the lake levels for this kind of events are predicted reasonably well. The eddy viscosity profile along verticals is parabolic in character in most of the cases. The velocity field did not show any circulation pattern in the horizontal plan as observed in the physical model and on some satellite photographs. The motion develops typical backflow profile in the vertical direction coinciding with that of the wind. In addition, the vertical distributions are quite the same throughout different segments of the lake. All these features are most probably due to the limited number of grid points since details in lake geometry may significant influence the model results only if the number of computational points is increased. Another explanation is the failure to include the effects of mountains around the lake, which locally deflect the wind.

The model execution time can be decreased substantially when simplifying the model equations. As demonstrated by Harleman and Shanahan (1980), the Rossby number is small in Lake Balaton (0.02-0.2), which allows the neglect of convective terms. Computationally, this brings considerable savings, because depth integrated velocity fields and the vertical distribution of horizontal velocities can be evaluated in two separate steps. As in Kvon's model, the horizontal eddy viscosity may be neglected, too, because the horizontal Ekman number is very small. The shallowness of the lake also permits the elimination of the vertical velocities when determining the horizontal flow field.

With these approximations, one may derive the simplest, but still realistic, three-dimensional model called the Ekman-type model. A sophisticated computer program of an Ekman-type model was developed by Young and Liggett(1977) and made available to IIASA. It was adapted by I. Fisher (1980) for Lake Balaton. The model assumes constant eddy viscosity throughout the lake depth. The solution is based on the finite element method which possesses the attractive ability to approximate accurately the irregular boundary of the lake shoreline. The computations are performed in the Laplace domain, and velocity profiles in real-time must be calculated by numerical Laplace inversion. After eliminating some numerical difficulties the model was used as before for a stepfunction wind only. On the basis of calculations performed (with the 200-250 elements limited by the PDP 11 computer) the same conclusion can be drawn as mentioned previously in connection with Kvon's approach: again, an essential increase in the number of space elements is needed, especially in the near-shore region (Fisher 1980).

In conclusion, the following may be stated about the use of three-dimensional hydrodynamical models for Lake Balaton. The models must be calibrated against measurement data by changing such unknown parameters as the wind drag coefficient, surface and bottom roughness, friction coefficient and the eddy viscosity and its distribution along depth and time. Because of the basically transient wind conditions (see 1.3) this procedure would need an unsteady approach. A well formulated Ekman model could be applied successfully for this kind of situation, and in fact, cooperation was initiated with the Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology (USA) in this respect. In connection with the calibration, the extreme importance of further lake current measurements must be emphasized as an addition to the existing data.

The basic question--which are the dominant water motion mechanisms (horizontal circulation, vertical backflow or longitudinal seiche) from the point of view of mass transport--can be answered only after calibrating the model. At the same time more can be said about the possibilities for simplifying the description of water motion to assure a necessary and sufficient coupling to biochemical processes.

3.2. Sediment-Water Interaction Models

As illustrated in Section 1, erosion of bottom deposits by wind induced effects is a mechanism of prime importance determining the suspended solid and particulate phosphorus concentration, as well as light transparency properties. Recent measurements in the Szemes Bay of total suspended solids, total phosphorus, ortho-phosphorus, and secchi disk depth on a daily basis, together with hourly wind data provide a considerable understanding of the processes of erosion and deposition (Somlyódy 1980). In a first analysis of the data starting with an unsteady transport equation for suspended solids and some energy transformation principles, Somlyódy (1980) derived a relatively simple model to describe the dynamic variation of depth-averaged SS concentration in time. The model used is:

$$\frac{d\bar{c}}{dt} = -k_1\bar{c} + k_2W^m \quad , \tag{1}$$

where $\bar{c}$ is the depth-averaged suspended solids concentration, and W the wind velocity at 10 m above the lake surface. Thus, the sedimentation is supposed to be proportional to the concentration whereas resuspension depends solely upon the wind. In the range of wind speeds 0-45 km/h, corresponding to a suspended solids range from 8-53 mg/l, the coefficients were found to be $m \approx 1$, $k_1 = 1.5 * 10^{-5}\ s^{-1}$ and $k_2 = 0.91 * 10^{-7}$ kg m^{-4}. Here,

k_1 is inversely proportional to the depth H, while k_2 is proportional to H^{-2}. It should be noted that at the given depth, H = 4.3 m, the value of k_1 corresponds to a sedimentation velocity of 5-6 m/day, a realistic value.

Given the suspended solids concentration, the secchi disk depth can be computed. Reworking the data reveals that in the range indicated the following correlation holds approximately:

$$Z_s = \frac{14}{\bar{c} + 4.9} \quad , \tag{2}$$

where Z_s is the secchi disk depth (van Straten 1980). Also a fairly strong correlation between total phosphorus and suspended solids concentration was found (Somlyódy 1980).

Unfortunately, the results presented in the previous paragraph do not allow for clear conclusions about the release of dissolved phosphate to the lake. The top layer of the sediment is probably very loose, with interstitial dissolved phosphate concentrations comparable to those in the lake. Release due to resuspension is then mostly dependent upon diffusion from deeper layers. To date no suitable experimental results exist which would permit the design of a comprehensive dynamic phosphorus release model. Data characterizing the chemical composition of the sediemnt layers, both of the solid part as well as the interstitial water do exist (Dobolyi 1980), but dynamic diffusion data, including the very important oxygen dynamics, are lacking. On the average, Dobolyi's data show an interstitial ortho-phosphate concentration of roughly 80 mg/m^3 in the top 5 cm, increasing to about 140 mg/m^3 between 20-25 cm. The total dissolved phosphorus content was virtually constant at roughly 200 mg/m^3. These data were gathered during July and September, 1978, and nothing is known about the time variability.

3.3. Lake Eutrophication Models

At present three basic models exist for the investigation of the phytoplankton dynamics in the lake; BEM, BALSECT, and

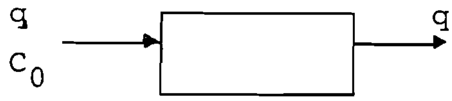
SIMBAL. The BEM model is the earliest model and was developed by the Hungarian Balaton Ecological Modelers Group. Various versions of this model have been used, but only the published version will be mentioned here (see for background: Herodek and Csaki 1980; for equations: Csaki and Kutas 1980; and for results Kutas and Herodek 1980). Emphasis in this model lies on the description of phytoplankton biomass and daily primary production. The model BALSECT (Balaton Sector Model) was developed at IIASA from an earlier more general phosphorus and nitrogen cycling model. It concentrates on the analysis of the models of phosphorus compound transformation in the lake (Leonov 1980). The third model, SIMBAL (Simple Balaton Model), actually consists of a set of models designed especially to investigate various hypotheses about the modes of phosphorus interaction between water and sediment, while explicitly taking the uncertainty in the data into account (van Straten 1980).

In the sequel the models are compared in various aspects such as: approximation to the spatial mass transport, type and number of state variables, number of forcing functions and parameters, and mathematical formulation of major processes. Finally some results will be discussed.

Transport. Although the bio-chemical and chemical transformation processes are the major theme of all three models, the spatial mass transport must be included in harmony with its role in ecology. Furthermore, a comparison with field data would be impossible without somehow taking the effects of water motion into account. Since a comprehensive analysis as mentioned in section 3.1. was not yet available, different ad hoc solutions were implemented in the three models. These solutions are summarized in Figure 9.

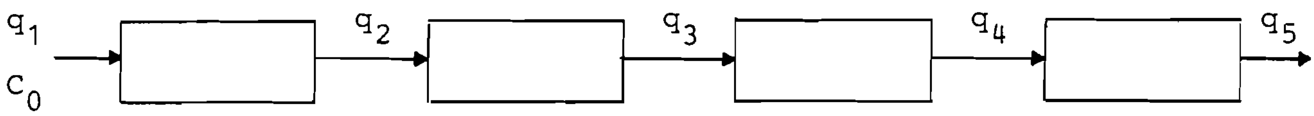
The transport representation in all three models is based on the four-basin segmentation of the lake as described previously. However, the BEM-model only operates for one basin at a time, without interconnection. The inflow and outflow are time invariant. In the BALSECT model only net hydrological transport is allowed for, where the transport terms are derived from the monthly water balance data. This is done in SIMBAL

BEM



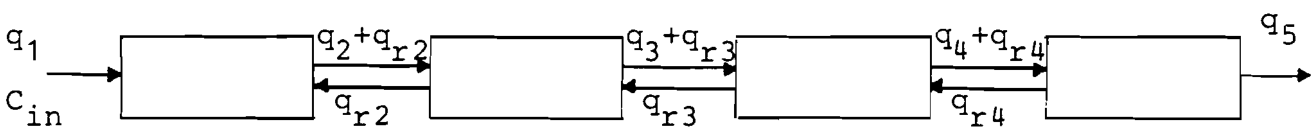
$$V\dot{C} = qC_0 - qC + RV$$

BALSECT



$$V_i\dot{C}_i = q_i c_{i-1} - q_{i+1} c_i + RV_i$$

SIMBAL



$$V_i\dot{C}_i = (q_i + q_{ri})C_{i-1} - (q_{i+1} + q_{ri+1})C_i$$

$$- q_{ri}C_i + q_{ri+1}C_{i+1} + RV_i$$

$q_{ri} = V_r A_i$

$q_{r1} = q_{r5} = 0$

V_r = exchange velocity (m/s)

A_i = intersegment cross section area

$C_5 = 0$

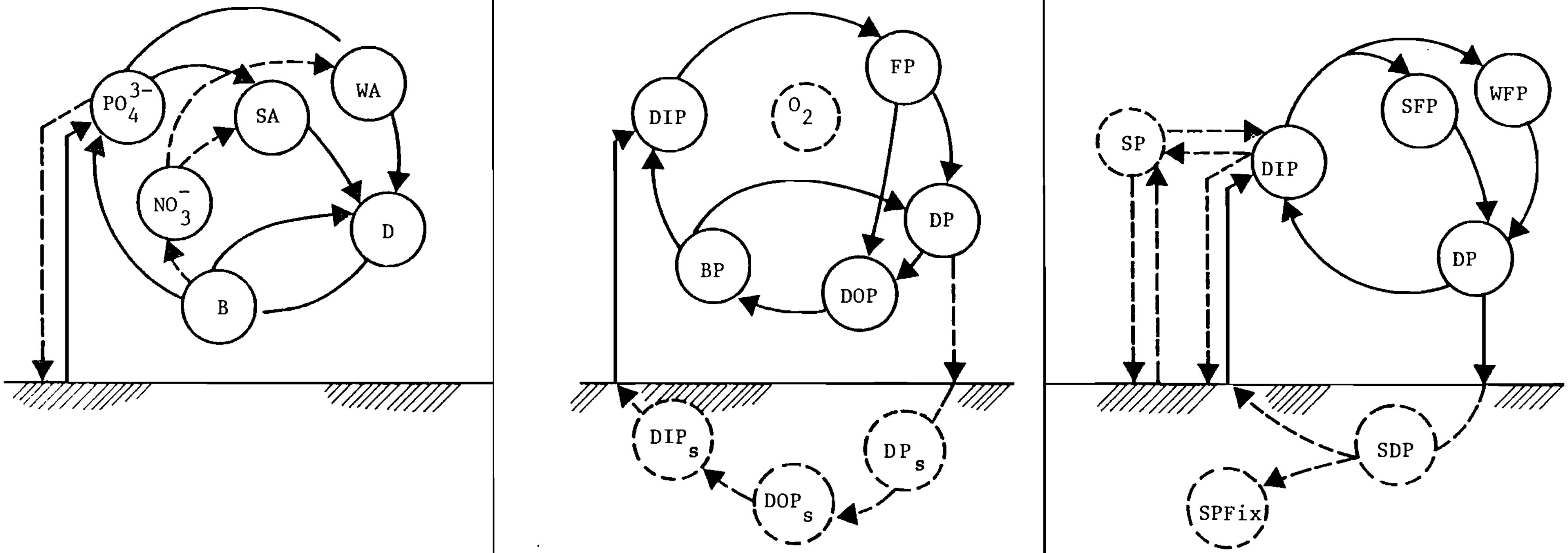
Figure 9. Comparison of spatial transport process descriptions.

too, but in addition allowance is made for the inclusion of wind-induced exchange flows, though presently as a non-dynamic term only. This construction is not fully equivalent to the integrated hydrodynamic transport models discussed in section 3.2., but it is considered at present to be sufficient given the uncertainty in both ecological and hydrodynamical data.

The different presentations of the transport process will, of course, effect the outcome of the model simulations. Obviously, the models have to be unified in this respect before comparisons of the performance of the biological and chemical segments among the models can be made.

Structure. A schematic overview of the structure of the models is presented in Figure 10. All three models show the cyclic structure typical for phytoplankton dynamics. Algae grow under uptake of nutrients. Mortality processes produce dead organic matter (detritus) which is subsequently mineralized to nutrients, thus closing the cycle. Generally, there is a fractional loss of nutrients to the sediment during each cycle. The differences in the biochemical structure of the models lie mainly in the number of state variables considered, and the mathematical description of the major processes.

In the BEM-model both nitrate and phosphate are considered as limiting nutrients. However, nitrate is limiting occasionally in the Keszthely Bay only. The model specifies summer and winter algae as separate state variables. The equations are identical but maximum growth rate, mortality and temperature dependencies are different. Accounting for these seasonal differences in algal composition is also possible in BALSECT, where seasonally different parameters and temperature dependencies can optionally be specified. In SIMBAL, a two-peak temperature function derived from primary production data analysis was used originally to account for population differences, but later two algal species groups were distinguished.



Mixed biomass-nutrient model

SA : summer algae biomass
WA : winter algae biomass
D : detritus concentration
B : bacteria biomass
PO_4^{3-} : ortho-phosphate concentration

NO_3^- : nitrate concentration

Phosphorus cycle model

FP : phytoplankton P
DP : detritus P (particulate)
DOP : dissolved organic P
BP : bacteria P
DIP : dissolved inorganic P

O_2 : oxygen concentration (not shown)
DP_s : detritus part.P in sediment
DOP_s : diss.org.P in sediment
DIP_s : diss.inorg.P in sediment

Phosphorus cycle model

SFP : summer phytoplankton P
WFP : winter phytoplankton P
DP : detritus P (both dissolved and particulate)
DIP : dissolved inorganic P
SF : sorbed P

SDP : sediment detritus P
SPFix : non-available sediment P

Figure 10. Comparison of stucture of major Balaton phytoplankton dynamics models.

The major diffences of the models lie in the algal decay and remineralization processes formulation. The most complete model in this respect is BALSECT, where a distinction is made between mortality (both of phytoplankton and bacteria) which leads to particulate dead organic material (detritus), and excretion. Excretion by phytoplankton is assumed to produce dissolved organic phosphorus, whereas excretion by bacteria regenerates the dissolved inorganic phosphate. Bacteria are also explicitly specified in the BEM model as the major agent of decomposition of detritus material. In the BEM model no distinction is made between particulate and dissolved detritus forms. Also, no sedimentation of detritus is included. In SIMBAL, again no distinction is made between the various detritus forms. Furthermore, the bacterial processes are dropped on the grounds that no measurements for this state variable are available. Instead, a time-varying decomposition (or remineralization) rate is obtained by introducing a strong temperature dependency, which is hope to reflect reasonably well the effects of a dynamic bacterial population.

The basic equations for the three models are presented in Table 4. For clarity, non-essential state variables have not been included: nitrate is omitted from BEM and oxygen from BALSECT.

All three models are constant stoichiometry models, i.e. the phosphorus (and nitrogen) content of the algal cells is constant. In the BEM model this ratio must be stated explicitly (x in Table 3). This is not necessary in BALSECT and SIMBAL because they are phosphorus cycle models. However, nutrient uptake and algal growth are supposed to occur simultaneously, hence constant stoichiometry is implicitly assured. It should also be noted that for comparison with biomass or chlorophyll data the ratio phosphorus/biomass or phosphorus/chlorophyll must be known.

Table 3. Basic Equations.

Model	Component	Symbol	Algal: Growth/uptake	Algal: Mortality	Algal: Excretion	Bacterial: Growth	Bacterial: Mortality	Bacterial: Excretion	decay/mineralization	Sedimentation	Sediment release/sorption
BEM	Algae	A	$+k_{ga} f_a A$	$-k_{ma} A$							
BEM	Detritus	D		$+k_{ma} A$		$-k_{gb} f_b B$	$+k_{mb} B$				
BEM	Bacteria	B				$+k_{gb} f_b B$	$-k_{mb} B$	$-k_{eb} B$			
BEM	Phosphate	P	$-xk_{ga} f_a A$					$+xk_{eb} B$		$-k_g Pk_{ga} f_a A$	$+k_{rel} P_s$
BALSECT	Algae	FP	$+k_{ga} f_a FP$	$-M_f FP$	$-L_F FP$						
BALSECT	Detritus	DP		$+M_F FP$			$+M_B BP$		$-k_d DP$	$-S_i DP$	
BALSECT	DOP	DOP			$+L_F FP$	$-k_{gb} f_b BP$			$+k_d DP$		
BALSECT	Bacteria	BP				$+k_{gb} f_b BP$	$-M_B BP$	$-L_B BP$			
BALSECT	DIP	DIP	$-k_{ga} f_a FP$					$+L_B BP$			$+k_{rel} P_s$
SIMBAL	Algae	FP	$+k_{ga} f_a FP$	$-k_{ma} FP$							
SIMBAL	Detritus	DP		$+k_{ma} FP$					$-k_d DP$	$-S_i DP$	
SIMBAL	DIP	DIP	$-k_{ga} f_a FP$						$+k_d DP$	$-k_s DPK_{ga} f_a FP$	$+k_{rel} P_s$ $[+k_{sor}(DP-DP_{eg})]$

Table 4. Combined Nutrient and Light Limitation for the Three Models: N, P, FP is nitrogen, phosphorus and algae-phosphorus concentration respectively

BEM	$f_a = (F_I \cdot FP \cdot F_N)^{F_O} \,.\, F_O^{\,1-F_O}$
	$F_O = \min\,(F_I,\ FP,\ F_N)$
	$FP = \dfrac{P}{P_K + P}$
	$F_N = \dfrac{N}{N_K + N}$
BALSECT	$f_a = F_I \cdot FP$
	$FP = \dfrac{P}{\frac{1}{\beta}\, FP + P}$
SIMBAL	$f_a = F_I \cdot FP$
	$FP = \dfrac{P}{P_K + P}$

Notes:

k_{ga}, k_{gb} - temperature dependent maximum growth rate for algae and bacteria respectively ;

f_a, f_b - light and nutrient limiting factors for algae and bacteria respectively ;

k_{ma} k_m - temperature dependent mortality rate for algae and bacteria respectively ;

k_{eb} - temperature dependent excretion rate ;

x - phosphorus-biomass ratio ;

M_F, M_B - temperature and uptake dependent mortality for algae and bacteria respectively (see text) ;

L_F, M_B - temperature and uptake dependent excretion for algae and bacteria respectively (see text);

k_d - detritus decay or mineralization rate;

k_s - biogeneous lime coprecipitation rate;

S_i - net sedimentation rate;

$k_{rel}P_S$ - effective lumped sediment phosphorus release;

k_{sor} - adsorption/desorption rate (SIMBAL II only);

DP_{eg} - equilibrium concentration dissolved inorganic P (SIMBAL II only).

Light Limitation. All three models use the Steele equation for the dependence of algal growth on light, which allows for inhibition at high light intensities as observed in the lake. The depth averaged equation is:

$$F_I = \frac{e}{k_e H}[\exp(-RX) - \exp(-RI)] \quad , \tag{3}$$

where

k_e = extinction coefficient

H = depth

RI = I/I_s

I = light intensity at the surface

I_s = optimal (saturation) light intensity

RX = RI $\exp(-k_e H)$.

In BALSECT this factor is evaluated at every time step using a sine-wave approximation for the daily light variation. Although this procedure is the most correct one, computation time can be saved by integrating this expression over 24 hours and using the result as a value constant over the day. This procedure is used in SIMBAL, based on a triangular approximation of the light variation over the day. The BEM-model employs the light function without integration, so that the daily averaged light intensity must be used in F_I, a somewhat less accurate procedure.

Nutrient Limitation. The models differ considerably in the description of nutrient limitation, as shown in Table 4. SIMBAL uses the classical Michaelis-Menten approach. This is also done in BEM, but instead of multiplying the nutrient and light limiting terms the minimum is calculated (and used in the model as an option), and a geometric mean of product and minimum is used as the combined effect factor. BALSECT differs essentially from the other two models. The Michaelis-Menten expression with constant half-saturation constant is considered not fully satisfactory in describing nutrient limitation. By analogy with bacteria processes, Leonov (1980) has replaced the half-saturation constant by the phytoplankton phosphorus concentration. In this way a lower threshold limiting concentration is reached at lower biomass than at higher biomass. The factor β is a parameter of the order 1.

Mortality and excretion. Here BEM and SIMBAL assume simple temperature dependent first order processes. Again BALSECT differs considerably. Excretion of dissolved organic phosphorus is assumed to be proportional with phosphorus uptake by growth

$$L_F = r_F \cdot K_{ga} f_a \quad , \tag{4}$$

where the proportionality factor is again a complicated two-parameter function of the uptake rate (see Leonov 1980).

The mortality rate is thought to be proportional to the biomass-uptake rate ratio:

$$M_F = V_1 \frac{FP}{k_{ga} f_a} \quad . \tag{5}$$

Thus, a low uptake activity at high biomass causes a high mortality, and in this sense the uptake rate biomass ratio is a measure of viability. Note that expression (5) leads to a second order equation for mortality (which is consistent with the logistic growth equation frequently used to describe a batch laboratory experiment).

Bacterial Uptake. Here BEM and BALSECT use the same expression for the nutrient limitation. Instead of Michaelis-Menten kinetics the same construct is used as described for the phytoplankton growth:

$$\text{BEM} \qquad f_b = \frac{D}{B + D} \quad , \tag{6}$$

$$\text{BALSECT} \quad f_b = \frac{DOP}{BP + DOP} \quad , \tag{7}$$

i.e. the bacterial biomass or bacterial phosphorus concentration replace the half-saturation constant used in the Michaelis-Menten equation. Mortality and excretion are treated in the same fashion as described for phytoplankton.

Biogenic Lime Co-precipitation. Both BEM and SIMBAL allow for the possibility of phosphorus removal due to co-precipitation with biogeneously generated calcium-carbonate during periods of strong primary production. However, recent measurements (Dobolyi and Herodek, 1980) suggest that the mechanism is not very significant despite the favorable pH (8.1-8.3).

Release From the Sediment. Although various models do contain a dynamic sediment section (BALSECT, SIMBAL, BEM later versions) lack of data does not permit the use of these sections. Rather a constant or temperature dependent release rate is postulated. In addition, the most recent version of SIMBAL allows for a dynamic adsortion/desorption process, such that desorption (i.e. release) occurs when dissolved phosphorus concentrations are low (during algal bloom), and adsoption when orthosphosphate is high (in winter). This hypothesis was introduced in response to results obtained with the first version of SIMBAL (see below).

Temperature Functions. Nearly all rate constants are functions of temperature. This is important in Lake Balaton because the temperature ranges from 0-26^{o}C. Generally, decay, mortality and excretion temperature functions are formulated either as linear or as some form of exponential expression. Temperature functions for the growth of the biological components are more complex: mostly bell-shaped curves are used to enable the specification of an optimum and a lethal temperature. The general equation is:

$$F_T = \max[0, \frac{TCRIT - T}{TCRIT - TOPT} \exp(1 - \frac{TCRIT - T}{TCRIT - TOPT})] \qquad (8)$$

where TCRIT is the lethal temperature and TOPT the optimal temperature. For further details the reader is referred to the original publications.

Number of Parameters and Forcing Functions. Before any model can be run, a number of numerical data must be transferred to the model. This includes first of all geometrical data, such as volume, depth, cross-sectional area. These are given in van Staten et al. (1979) and are the same for all models. Forcing functions must also be specified. These are daily radiation, photo-period, water temperature, and phosphorus inputs. Unfortunately, not the same loading assumptions have been adopted in the three models. Particularly in the BEM and BALSECT models, additional inputs had to be assumed in certain periods of the year to improve the fit, without any experimental justification being available. Some of the parameters for which a submodel could have been used are also introduced as forcing function tables or constants. These are the basic extinction coefficient and, in SIMBAL, the return flow.

The number of parameters that must be specified greatly differ among the models. To enable a comparison, in this respect, Table 5 summarizes the parameter needs, classified in various groups. Non-essential state-variables have been removed (such as nitrate and oxygen), but the two algal species were retained in all models (the additional cost of increasing the number of algae species is also indicated).

As can be seen from the table by far the most demanding are the temperature functions. However, it is likely that the models are not very sensitive to the majority of these parameters. Even if the temperature functions would be completely known the number of parameters to be estimated is still relatively high: 15 for BEM, 17 for BALSECT, and 12 for SIMBAL. It should also be noted that, in principle, these parameters may be different for different parts of the lake.

Table 5. Number of state variables versus number of parameters (in brackets: increment for each additional algal species).

	BEM	BALSECT	SIMBAL
Essential State Variables	5 (1)	5 (1)	3 (1)
Parameters			
Basic Rate Constants	9 (2)	12 (4)	6 (2)
Nutrient Limitation Factors	2 (1)	1 (1)	1 (1)
Light Limitation Factors	3 (1)	3 (1)	3 (1)
Temperature Functions	14 (2)	16 (2)	11 (2)
Stoichiometric ratio	1	1*	1*
Return flow	-	-	1

*implicit, see text.

Results. Results obtained from simulation studies with the models described above have primarily contributed to a better understanding of the complex phenomena in the lake, and were very important in indicating improvements in both measurement and modeling techniques. The models prompted special experiments to investigate the principle modes of phosphorus removal from the lake water. A result common to all simulations is that at the given loading rate appreciable amounts of phosphorus are transferred to the sediment and that the direct release of phosphorus still plays a minor role (with the exception of the possible role of adsorption/desorption in the annual dynamics, see below.

The BEM model has been used only to simulate one basin at a time, and was concentrated on simulation of biomass and primary production. As shown in Figure 11 for the Szemes Basin (basin III), the primary production is reasonably well represented, except for a period in autumn. The spring peak could only be

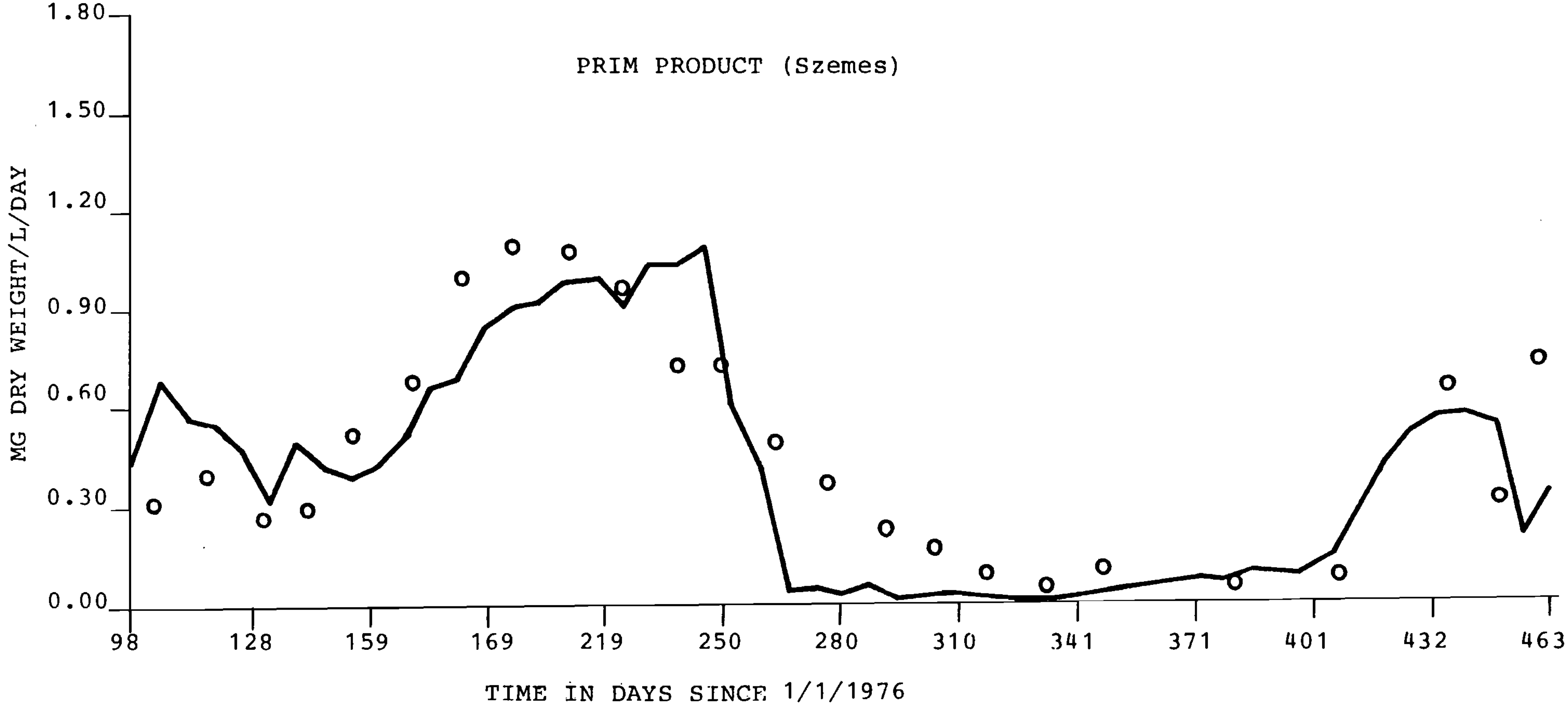


Figure 11. Results of BEM simulation for Szemes Basin 1976. Primary production.

simulated if the assumption was made that the lake received an additional loading of organic material after the snow melt. The fit was also quite good for biomass, but still rather poor for orthophosphate. Here the model predicts too much phosphorus at the time when the algal activity declined, a tendency that can be observed in all models. The question may arise as to whether in spring and winter alternative phosphorus removal mechanisms are operative, for instance due to benthic algal activity. Only measurments could help to clarify this issue. Simulation for Szigliget Bay did not yield as good results. This may be partly due to the presence of blue-green algae in the real systems. None of the models contains a mathematical description allowing for a simulation of blue-green algae. It should be noted that the BEM group made no attempt to compare the BEM model results with the extensive phosphorus, nitrogen, and chlorophyll data available for the lake, although it would certainly increase the reliability of model calibration. Furthermore, the urgent need to extending the model to the whole lake should be emphasized.

An impression of the relative importance of the phosphorus transformation processes is given in Figure 12 obtained with BALSECT for Keszthely Bay in 1977. The hydrological throughflow is insignificant as compared to the external loading and the internal cycle. According to the model, mortality is a more significant mechanism of phosphorus cycling than excretion, but contributes at the same time most to the loss. It is also apparent from this figure that the Zala load dominates the phosphorus simulation in Keszthely Bay. An impression of the match between data and simulation results is given in Figure 13. Given the uncertainty in the data this result is reasonable, and an attempt for a more sophisticated fit is probably not worthwhile. It should, however, be repeated that assumptions made in the model about the distribution of the loadings over the months of the year, are not supported by experimental evidence. Also, the sedimentation rate was varied from month to month in order to improve the fit. In view of both the uncertainties in model and data, it is doubtful that a reliable time pattern can be identified in this way. Later, although, an attempt was

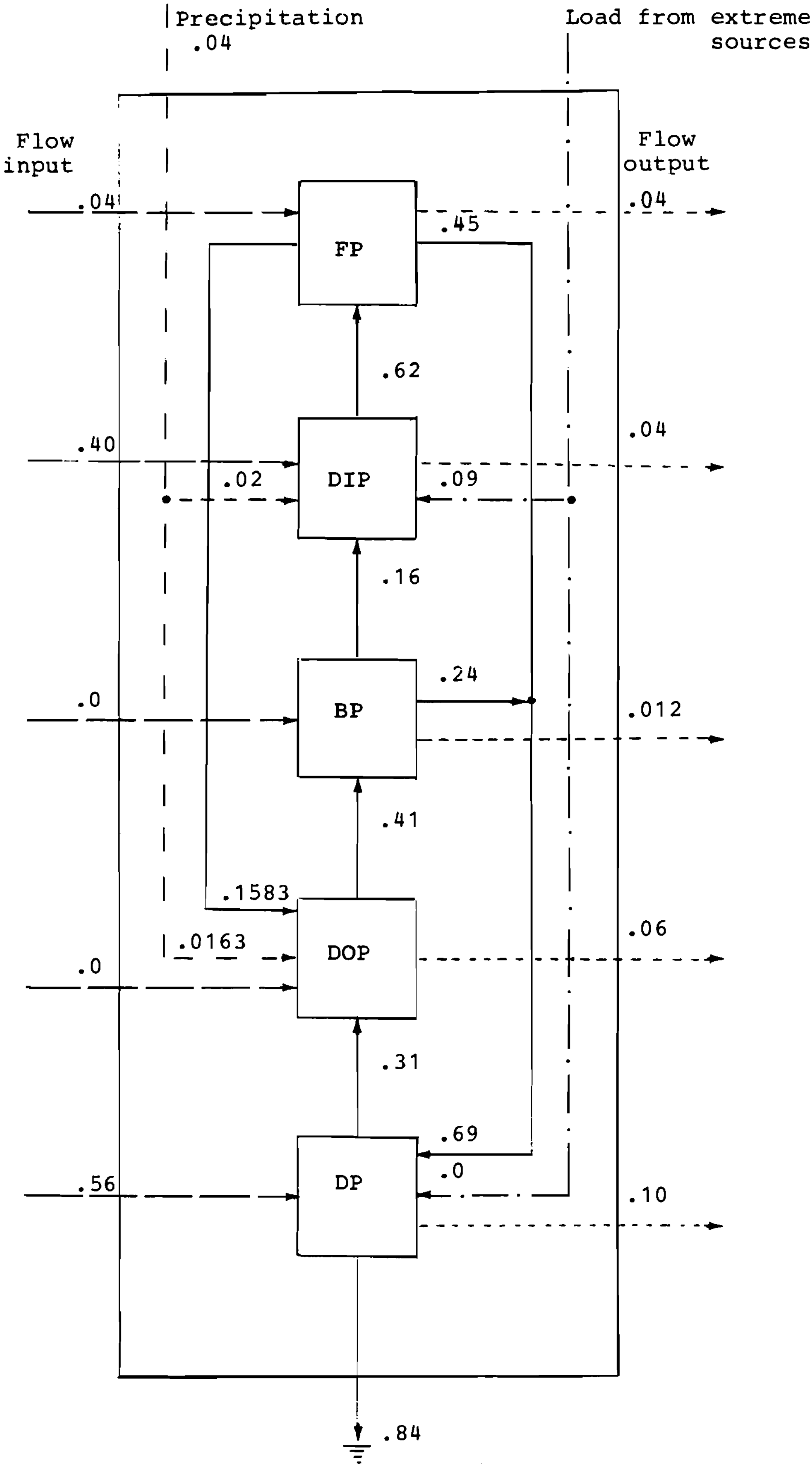


Figure 12. Results of BALSECT phosphorus transformation flows for Keszthely Bay, 1977 (in mg/1, yr.).

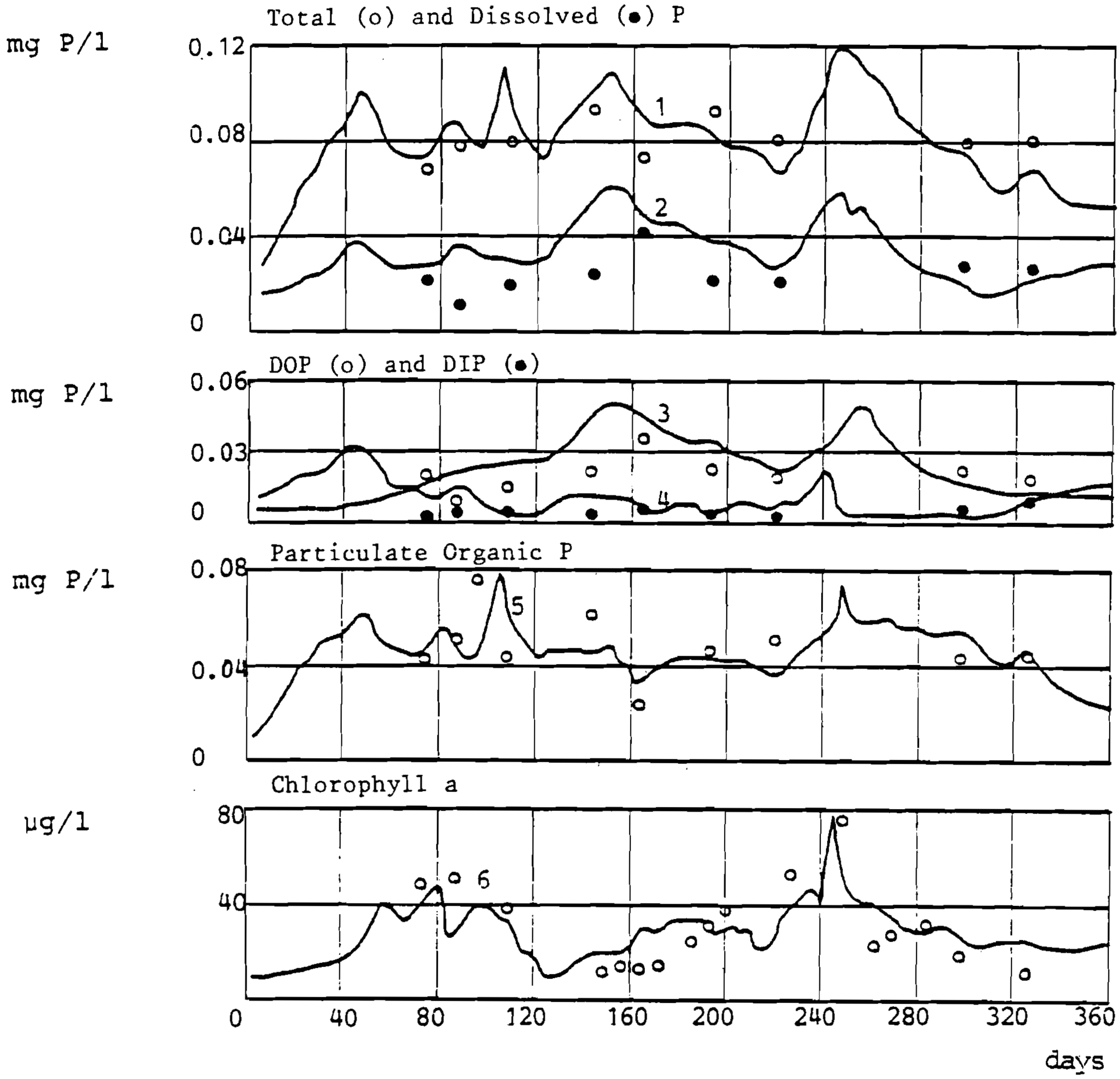


Figure 13. Results obtained from BALSECT. Comparison between model output (curves) and field data (points), Lake Balaton, 1977, Keszthely Bay (hypothesis of three phytoplankton groups).

made to correlate the monthly variable sedimentation rate to some wind characteristics. It is, therefore, more appropriate to derive net sedimentation rates from the use of the resuspension submodel described earlier. It is also important to investigate the effect of longitudinal mixing. First results with SIMBAL suggest that the latter can be extremely significant under strong wind conditions.

The use of SIMBAL differed from the other two models, because SIMBAL was primarily designed to test hypotheses about the modes of phosphorus transformation in the lake. The procedure is such that the model performance for a randomly selected parameter combination is compared with a behavior defined for the system from the data rather than with the data directly. In this way allowance is made for uncertainty in the data explicitly. It turned out that with a first version of the model (without the sorption mechanism) no parameter combination could be found that produced the desired model behavior. (In fact, too much ortho-phosphate was predicted in autumn, a common problem to all models as mentioned before). This led to the postulation of the sorption hypothesis, and in this case parameter sets could be found that produced the defined behavior. Since there is more than one parameter set, the simulation result constitutes a probability distribtion around the mean time trajectory rather than one single deterministic time trajectory. This is illustrated in Figure 14 for phytoplankton. Although the result is not unreasonable (for a detailed discussion see van Straten, 1980), several other hypotheses may be tested using the same procedure. In the present stage of development, the methodology provided is more important than the actual results. The method allows the determination of whether a certain hypothesis is consistent with the data or not, though the true confirmation can only come from experimentation in the field.

4. NUTRIENT LOADING

Although several nutrient loading models do exist in the literature, and are, in fact, studied by REN's task on Environmental Problems of Agriculture, lack of data has hindered

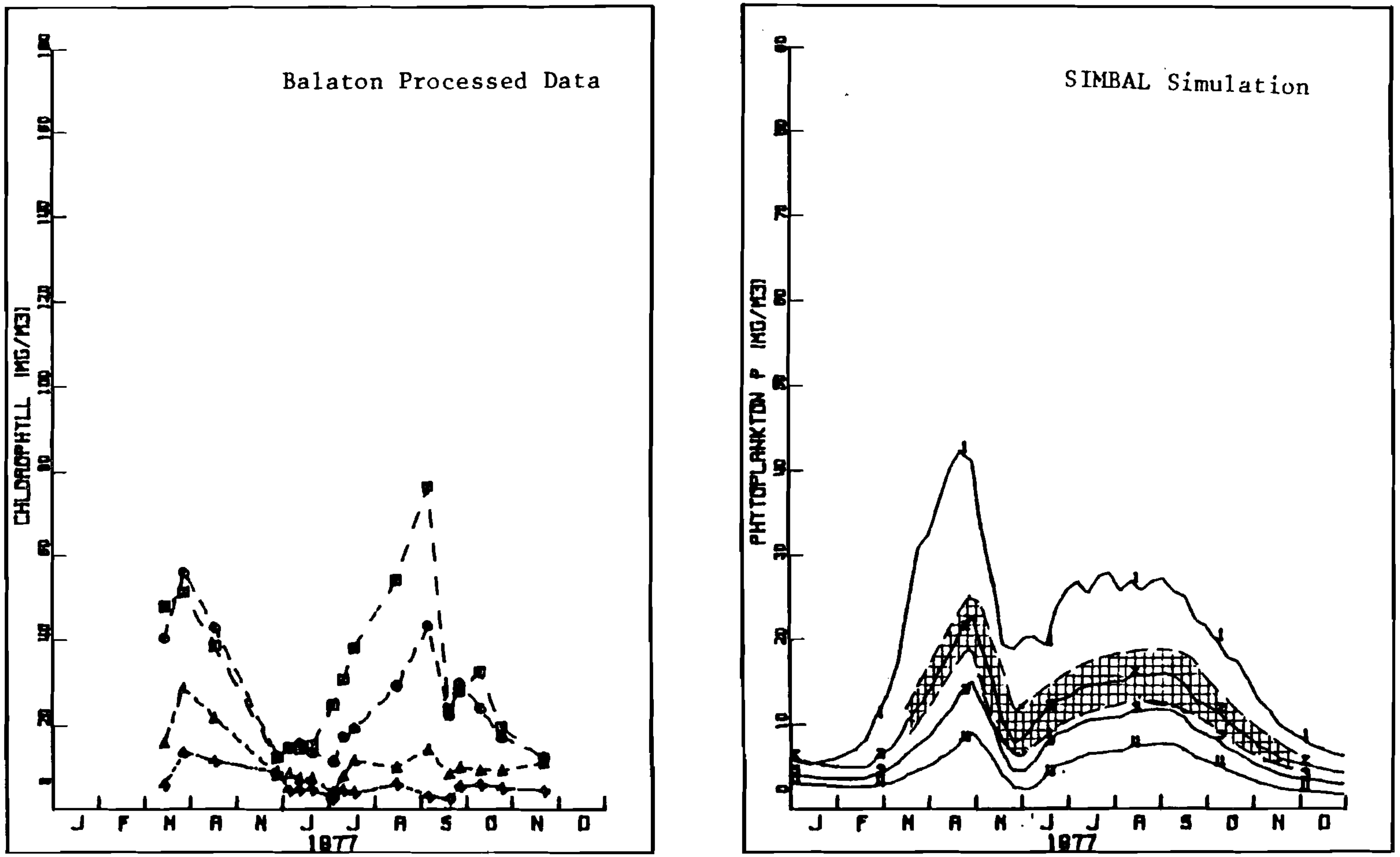


Figure 14. Results from SIMBAL. Comparison of field data for the four basins (left) and average of model simulations for the behavior producing parameter sets (right). For illustration the estimated standard deviation around the trajectory for Basin 2 is also indicated.

application for Lake Balaton so far. Most of these models require a fair amount of geographical, geomorphological, geophysical, and agricultural data in conjunction with stream discharge and nutrient load information. Although discharge and nutrient load measurements are part of the regular measurement scheme of the Water Authorities, much of the loading occurs during rainfall events, which are normally not sampled. Only one event-based field study is known, namely for the Tetves-basin (70 km^2) (Jolánkai 1977). Once the associated geographical data (such as slope conditions, soil properties, land use types, etc.) have been compiled, the results of this study and a similar study underway at present, can be a valuable basis for the application of one of the existing nutrient loading models.

In the meantime a less ambitious input model has been designed on the basis of the Tetves data. Basically, existing simple rainfall runoff relations were supplemented with a two-parameter phosphorus loading hypothesis. The input model starts from a rainfall event characterized by effective rainfall depth and rainfall duration. Runoff volume is estimated from the rainfall depth with the U.S. Soil Conservation Services formula (SCS). Similarly, sediment yield is estimated by combining the SCS formula and Universal Soil Loss Equation. Thus, the variables of runoff volume and sediment yield characterize the event. Nutrient loadings are then obtained by multiplying the runoff volume with some characteristic dissolved phosphorus concentration to obtain the dissolved P load per event, and similarly for fixed P by multiplying sediment yield with a fixed P concentration per unit sediment. By combining the events (and, if desired, watersheds), the seasonal nutrient loading can be obtained, provided that the characteristic P-concentrations are known, for example from measurements. In the case of the Tetves study these values were estimated by calibration of the model. Taking the variability per event into account it was found that a considerable loading variability may be expected from season to season. A version of the model where the events can be characterized by their stochastic properties (resulting in a probability density function for the seasonal load) is available at IIASA (Bogárdi and Duckstein 1978). It should be noted, however, that the crucial problem in the application of this model lies in the specification of the characteristic phosphorus concentrations. Before application in the management context can be achieved, a relationship between human activities in the catchment area and the model parameter must be found.

In recent years, detailed data have been collected for the Zala River, i.e. daily measurements of total phosphate and discharge at two measurement stations along the River. Since the Zala River sub-watershed is very large, use of these data for watershed models on the field level is probably not feasible. However, time series analysis might prove to be of interest.

5. MANAGEMENT MODELS

5.1. Watershed Development Approach

This approach to long-term management of the eutrophication problem was proposed by David et al. (1979), and is based on the hypothesis that a close relation exists between the human activity in the watershed and the degree of eutrophication in the adjacent waterbody. First a list was compiled of basic factors, which were believed to represent the impact on nutrient loading. There are three groups of factors:

-- natural factors (e.g. watershed area, average slope of arable land, length of rivers, etc.)
-- socio-economic factors (e.g. population, population working in industry, number of visitor-days, area of arable land, forest, vineyard, fertilizer use, etc.)
-- water management factors (e.g. actual total water consumption, domestic, industrial, agricultural water use, effluent collected in sewage works, treated effluent, storage reservoir capacity, etc.).

In total 50 factors were selected. The numerical values were evaluated from available data of large regional units, and separated over the watersheds belonging to the four basins of the Balaton as a preliminary step.

Next, the basic factors were combined into dimensionless indices, expressing the relative degree of development on a scale from minimum to maximum. The indices were defined in such a way that an increase represents an increase in nutrient loading to the lake. Examples of important indices are relative population density, average surface slope, density of point sources, fertilizer use index, ratio of untreated to treated sewage.

Once the system of indices had been established, a panel of experts working on the Balaton Case Study was asked to weight the various indices. The sum of the weighted indices provided one watershed development factor.

The results are summarized in Table 6.

Table 6. Preliminary results of watershed development approach study.

Basin*	I	II	III	IV
Development Index %	42	40	43	60
Relative Area	0.53	0.32	0.1	0.05
Nutrient Loading Index	22	13	4	3
Summer Maximum Biomass mg/1	8.5	5.0	2.8	2.3

* see Figure 1.

Although the Siófok region (basin IV) has the highest degree of development (with respect to nutrient loading) its contribution to the nutrient loading is small due to the very small surface area compared with the total watershed. The spatial distribution of the nutrient loading index correlates well with the distribution of summer maximum biomass, selected as a measure of eutrophication. Correlations with other eutrophication indexes were also investigated. The weighting system had only a limited effect on the correlations found.

The results of this first stage of the study are encouraging, but at present not sufficient for long-term planning. Of course, the availibility of data only for 1975 did not allow the identification of trends, and the use of historical data is certainly required. However, a more serious problem is the lack of sensitivity to how the various catchment area activities are weighted. This lack of sensitivity means that the nutrient load does not depend on the relative importance of each of the varying activities, which is probably not the case in reality.

5.2. Other Management Models

No publications exist yet on other work on management models performed in the framework of the case study. One study is in progress (Duckstein et al. 1980) and is worth mentioning here. The basis for this approach is the use of literature data on the relation between watershed activities and phosphorus losses. A set of realistic management alternatives, such as tertiary treatment, contour farming, changed tillage practice, reduced fertilizer use, sediment retention reservoirs, etc., and combinations, is formulated. The literature data are then used to provide the data for a stochastic input model (described below). This model generates the annual nutrient loading and the stochastic distribution for each alternative, for subsequent use in some kind of management alternative selection model. The first results are expected in the middle of 1980.

Other management oriented approaches will be developed in the near future. Here the need for detailed physical and economic data must be emphasized.

PART II. FUTURE PLANS

1. General
2. Lake Eutrophication Models
3. Nutrient Loading Models
4. Water Quality Management Models

1. GENERAL

1.1. Practical Needs of Research

The goals and objectives of the Case Study were summarized in Part 1 in coincidence with the agreement between IIASA and the Coordination Council for the Environmental Research on Lake Balaton. Mainly questions connected to the lake's ecology were addressed and the actual application of the models to be developed was mentioned only tangentionally. During the last two years, however, the urgent need to implement control activities in order to imporve the lake's water quality has become better recognized. This finding has two consequences, namely: (i) decision makers need the results of the study--assuming that they are available at the appropriate time in appropriate form--and (ii) there is a unique opportunity for researchers to have their results quickly applied in practice. Consequently, besides answering ecological questions, one has to concentrate more and more on solving the problem of management: how to elaborate and schedule control strategies, what are the expenses and how to find a realistic compromise among the different alternatives (see Figure 8).

1.2. Basic Principles

The reader is referred here to Figure 7 which illustrates the hierarchy of the different models. The ecological problems are addressed mainly by the Lake Eutrophication Model (LEM), while the "practical" problem (the improvement of the lake's water quality) has to be solved on the level of the Water Quality Management (WQMM) which comprises LEM and the Nutrient Loading Model (NLM) as submodels. Since WQMM should solve an optimatization problem, coupling of the various models is required, which is not a simple task. To illustrate the concept to be applied two interconnected but different steps will be distinguished in the use of LEM and NLM: (i) first, detailed simulations are performed primarily on a daily basis (see Part II), for calibration and validation of the model before using it for any prediction; (2) secondly, the model can be applied in a more global sense characterizing the large-scale and long-term behavior of the system as expressed by certain chosen indicators such as yearly peak values, different averages, duration of critical concentrations, frequency distributions, etc., which then should serve as a basis for lake management. The number and type of indicators must define unambigously the "global" behavior of the lake from the point of view of water use (here recreation is of primary interest).

In this line of thought the same model could serve both purposes. It is evident that the second type of model application (ii) is less sensitive to the subprocesses.

Given the final interest in application in a management framework one might suggest to strive directly for a simplified (static, semi-static, or linearized dynamic) model. However, in that case validation of the model is much more problematic, if not impossible, and there is also a chance that basic features (role of the sediment, for instance) are not grasped correctly. Therefore, a development from the detailed to the simplified with the help of an appropriate sensitivity analysis seems to be the only way to be followed.

This two-step procedure provides an off-line coupling of submodel results with the water quality management model. The reason to suggest this method is, that one has to be quite skeptical about the possibilities of direct coupling, knowing the complexities of the different subprocesses, the presence of many interactions among them, and the wide variety of timescales on which they operate. The off-line method is also one of the general tactics to be followed within LEM and NLM, for instance in coupling hydrodynamical models with the spatially and timely less detailed biochemical submodels.

The ideas outlined above would result in a consecutive development from LEM to a simplified LEM in the context of WQMM. This may lead, however, to a considerable time delay in the Case Study. As it is apparent from Chapter II.3., the models are at present only at a stage of calibration. In both LEM and NLM several unclarified questions have to be answered (see later) and several processes such as spatial mass transport and interaction between water and sediment have to be included more accurately. Accordingly, a successive development is not effective enough and therefore the simultaneous elaboration of models has been chosen. It allows at the present intermediate stage an early implementation of the complete model structure (see Figure 7) using the most up-to-date but not yet verified submodels for developing and testing methodologies rather than for solving the problem. This is an iterative type of procedure, the basis of which is the improvement of LEM and NLM up to the point of validation.

Three additional reasons give further support to this kind of development. In close connection with the modeling activities a data collection program and field experiments are under way. The approach explained above allows a similar iteration in this respect. Furthermore, after finalizing the project the same technique makes possible for the responsible Hungarian authorities to complete the background information and to use then the methodologies in a similar way before elaborating the final management alternatives for improving the lake's water quality. Lastly, the objective of the Case Study is not only to advance the solution of the present problem but to develop tools which can be applied for other shallow lakes as well.

A final remark is mentioned to conclude this section. As it could be learned from the proceeding part of this report, several models are being developed for the same problem. These are different in their background, structure, complexity, data requirements, deterministic or stochastic character, solution technique, etc. It is felt to be not only an important task but also a unique opportunity to compare these models under the same conditions. In the future primary emphasis will be given to this matter.

To summarize, the basic guidelines for our future work will be as follows:

(i) according to the progress in the Case Study and to practical needs there should be a greater concentration on WQMM;

(ii) an off-line principle is to be used in coupling submodels;

(iii) a simultaneous rather than sequential development of the different aspects of the study is to be followed, with iterative interactions among the models, between the models and data collection programs, and between the models and experimental work. It is not the intention to describe in great detail the subporcesses themselves, but rather to find the simplest decription (using sensitivity analysis) that is sufficient from the point of view of the process as a whole;

(iv) at this intermediate stage it is better to develop methodologies than to "solve" the problem with non-verified models;

(v) it will be highly desirable to compare different models for given purposes under the same conditions.

In the subsequent three chapters an overview will be given about future plans for the different aspects of modeling in the light of the basic ideas presented above.

2. LAKE EUTROPHICATION MODELS

2.1. Hydrodynamics

As discussed in Chapter II.3.1., we are interested here in the large scale spatial transport of different dissolved and particulate substances, in order to enable the coupling of the biochemical models of the different lake segments considered to be uniform from the point of view of water quality (see Figure 7). The basic question is which kind of water motion mechanisms (seiche, horizontal circulation, or vertical backflow) play the dominant role in the transport and how to overcome some of the modeling difficulties deriving from the structure of the governing equations.

Although Lake Balaton differs essentially from the Great Lakes in the light of experiences gained from the study (Halfon and Lam 1978, Lam and Halfon 1978, Boyce et al. 1979), it does not seem to be feasible to use a coupled more-dimensional hydrodynamic-transport model incorporating all biological and chemical reactions: gain in information is not proportional to the increase in complexity. Thus again an off-line procedure was chosen consisting of the simultaneous development of a one-dimensional (longitudinal) seiche model and a three-dimensional model, without direct coupling with biochemistry. The 3-D model is an Ekman type model in a version with vertically variable eddy viscosity (see II.3.1.). Using a refined grid, this model may answer the question of the dominant mechanisms of water motion (note that the horizontal circulation depends highly on the near shore conditions, the description of which by an Ekman model with linear bottom friction is not fully validated). The model may be calibrated against dynamic water level data and velocity measurements at the Tihany peninsula. The same can be done for the 1-D model and the results may be then compared. If the longitudianl seiche causes the dominant motion, as is thought to be the case, the two ouputs will agree reasonably well and the essentially simpler 1-D version can be satisfactorily used (additionally the role of non-uniformities in velocity found may be included into a dispersion coefficient used later on).

For the next step a one-dimensional transport equation will be coupled to the 1-D hydrodynamical model with the aim of studying first, the role of water motion on the mixing of a conservative material, and second, the importance of mixing related to physical and biochemical processes. In the course of the latter, biochemistry will be represented by simple reaction terms expressing time scales and the magnitude of changes correctly (e.g. some type of total phosphorus model). Here, real wind and loading data will be applied. The transport model will use the ouput of the 1-D seiche model for convection and the experiences of the 3-D version for dispersion.

The study will show whether the treatment of transport may be simplified further or not. For instance, if the wind influenced velocity has an oscillating character with zero mean value, the convective term can be neglected and a new, increased dispersion coefficient should be defined. As a next step the applicability of the completely mixed reactor concept with interchange as described in Chapter II.3.3. will be analyzed, which can be considered as the simplest way of solving the problem.

Depending on the final decision on the solution a direct relationship between the wind (inducing mass exchange) and parameter(s) appearing in the model (such as the newly defined dispersion coefficient) should be established. For this purpose a time series analysis may be helpful using measured wind data and calculated quantities (e.g. water velocity including its statistical properties for a certain period, dispersion coefficient).

In fact both models are currently under development. The main parameters for calibration are the eddy viscosity, drag coefficient, and bottom friction. It is particularly interesting that the 3-D model was found to be very sensitive not only to the average value of eddy viscosity but also to its distribution along depth. Changing only the latter, quite different seiche motions and velocity profiles were obtained. This fact calls for further study of the vertical eddy diffusivity, D. Here the application of two different methods is planned, namely:

(i) the use of Kvon's simplified, vertically one-dimensional model which is based on the k-ε turbulence theory (see II.3.1.) to provide an insight into D (W,z), where W is the wind speed, z is the vertical coordinate (Kvon 1977); and

(ii) estimation of D from the daily SS measurements at Szemes in five different points along a vertical (Somlyódy 1980). The details will be discussed together with sediment-water interaction modeling in the next chapter.

Although these two approaches may help considerably the importance of velocity and tracer measurements has to be emphasized once more.

For the solution of the 1-D problem an effective numerical method was developed which allows the economical use of the model for longer periods (e.g. a year). A thorough sensitivity and stability analysis and a comparision with analytical solutions for simplified conditions showed that the model behaves reasonably and accurately. The most striking feature is the important role of the spatial pattern of the wind (see Chapter II.1.3.) when applying the model to real situations. Since regular wind observations are available for two or three stations only, the uncertainty in wind distribution (magnitude and direction) has to be taken into account in future simulations. As transversal winds are also inducing longitudinal seiche motion, a correction procedure for the wind direction in the 1-D model is under elaboration using both experimental data and the conclusions from the 3-D model version.

2.2. Sediment-Water Interaction

The sediment layer acts as both a sink and a source of several dissolved and particulate materials. In the frame of the present study we are mainly interested in the sedimentation and resuspension of suspended solids and particulate phosphorus and in the release of orthophosphate P. All these processes are of major interest from the point of view of both the present state and future fate of the lake: SS influences the light conditions (see II.3.2.) and may play a controling role in the lake's water quality through the sorption mechanism (see II.3.3.), while the PO_4-P release works as an internal loading determining the new equilibrium and response time of the lake after reducing the external nutrient loading.

(i) In connection with the subjects outlined, it is principally the analysis of the daily measurements cited in Chapter II.3.2. that will be continued. In fact, the extended Kalman filter technique (see e.g. B. Beck 1979) for Equation (1) of that chapter is being applied for the total 6-month period. The first runs showed the model structure to be formulated well and gave parameters close to those estimated by a deterministic least squares fitting technique.

The SS model may also be used for calculating the extinction coefficient k_e by adding one more step to Equations (1) and (2) of Chapter II.3.2., i.e. the empirical relation between z_s and k_e (van Straten 1980). The model can then be applied to real situations for which wind data and measured extinction coefficients are available and the comparison may serve as a basis for verification.

(ii) Since phosphorus is found mainly to smaller particles its sedimentation and resuspension may differ from that of the suspended solids. It is planned, therefore, to extend the SS model for particulate phosphorus by using the P fraction data gained during the observation period.

Similar measurements with a daily sampling frequency are being taken at present in two basins of the lake: Keszthely and Szemes (phephytine is also included in the program, see I.1.6.). The data series will be included in the analysis outlined above and presumably will allow a generalization of the models developed for the whole lake.

(iii) On the basis of the SS data study one can estimate the pore water release as a function of wind. However, in translating this to an internal loading one encounters many difficulties, since the loading also depends on the PO_4-P concentration of the pore water. Due to lack of appropriate data we are not in the position to establish a dynamic sediment chemistry model (see Chapter II.3.2.). consequently, neither the orthophosphate concentration nor its temporal change can be calculated. Using Dobolyi's data for off-shore regions (which can be considered as a snapshot, only) the loading is quite small. Later measurements of Literáthy* with monthly sampling frequency in the near shore area, however, indicate concentrations that are not infrequently ten times higher and anaerobic conditions previal in most of the cases. Taking into account also the more effective release here due to the samller pathway in the wind energy transformation from the surface to the bottom a much higher loading (50-100 kg/d) can be estimated for the whole lake. This is approximately 10% of the dissolved external loading. In the course of a recent laboratory study (not yet completed), Herodek* found an even higher release of phosphorus (even though the overlying water

*Personal communication

was not stirred in the vessel). Finally it is worthy to mention that Gelencsêr and Szilâgyi* when collecting sediment with an Ekman sampler in the Szemes basin for sorption experiments (see later) found an PO_4-P concentration which would result in still somewhat ligher loading (note, moreover, that this concentration is a lower limit due to the sampling technique).

We may conclude here that even when assuming the wind induced relaease to be the dominant mechnism and its description to be well established one can not estimate reliably the internal loading which plays a crucial role from the point of view of management. Consequently, systematic measurements are urgently needed for allowing the estimation of at least lower and upper limits for the release. The role of time variability in the interstitial concentration is once more underlined and it is considered to be useful to perform regular and frequent measurements both in time and space besides making more detailed studies in a limited number of locations.

(iv) The sorption hypothesis of SIMBAL indicated how inportant a role the suspended solids may play in controling orthophosphate concentration. The same findings were obtained earlier by Olâh et al. (1977) on an experimental basis, however, no regular measurements are available for justifying this statement. Therefore, a detailed adsorption-desorption measurement program was worked out and is under development at VITUKI.

(v) To conclude this chapter we return to the estimation of the vertical eddy diffusivity as a function of wind and depth (see Section 2.2, item (ii)). The daily SS data measured at five different depths allows the application of the one-dimensional, transient transport equation to the problem with the help of Equation (1) of Chapter II.3.2. determining the boundary condition at the bottom. Thus it is possible to compute SS(z) and then to find D(z) or even D(W,z) by a trial and error procedure. First, promising results have

*Personal communication

already been gained by using an implicit finite difference solution algorithm. Many other alternatives are also of interest and under development such as:

-- Analytical solution of the same problem. Here the variables are separated such that an infinite number of ordinary differential equations results and among these presumably it is sufficient to solve only the first few equations of the series. The method assumes $D(W,z) = D(W)$.

-- The nice feature of the latter technique is that it allows a direct coupling with Kalman filtering and a simultaneous estimation of the boundary condition and $D(W)$.

-- The method of moments (Aris 1957) is also under application. In that case rather than calculating $SS(z)$ in details the first few moments of the same distribution will be considered, only, by transforming the original partial differential equation to a set of ordinary differential equations. The equation of the 0th moment is identical to Equation (1) in II.3.2., consequently, the inclusion of the next few (not more than four) moments into a Kalman filtering scheme is a natural extension of the original principle. This idea allows the simultaneous estimaion of k_1, k_2, m, $D(W)$, and additionally, the concentrations at the free surface and bottom, respectively.

Two more remarks are worth mentioning here. First, the development of different kind of methods permits not only comparison but also combination of the approaches (e.g. it is reasonable to use the $D(W)$ relationship obtained with the use of the Kalman filtering technique in the frame of the finite difference consideration and then to find the best fitting for $D(z)$ separately), and secondly, the present problem serves as an excellent example from the methodological point of view to identify and estimate the parameters of a system whose structure is governed by a partial differential equation.

2.3. Eutrophication Modeling

In this section the problems of biochemical and eutrophication modeling will be discussed (see Figure 7). The first of these has reached the most advanced stage among the different activities but is still far from complete understanding (see Part II).

With regard to further dievelopemnts the following comments may be given:

(i) In the light of the previous chapters, one step is obvious for improving the existing models: the coupling of hydrodynamical and interaction models to BEM, BALSECT, and SIMBAL in an identical fashion.

(ii) As indicated earlier, many questions may be raised concerning the lake's biology and chemistry. It suffices merely to remind the reader to the non-clarified role of blue-green algae and nitrogen limitation, the orthophosphate-P anomaly, the assumption of sorption, to the uncertainties in judging the relative importance of rate and capacity of simultaneous uptake mechanisms (uptake by algae, adsorption, and co-precipitation), the release of organic material and nutrients when benthic algae die off after ice-melting, the external organic material loading, etc.

When answering these questions one has to take into account that the uncertainties in the data and the limited knowledge of the relative importance of the various subprocesses listed--may well lead to several model versions having more or less the same "goodness". Consequently, it is not appropriate to fit models at every stage of development to field data, but rather it is suggested that these model hypotheses be tested first (depending on the execution time, e.g. in a similar way as it was realized in SIMBAL). One criterion in this procedure has to be emphasized: a hypothesis or a set of hypotheses for various subprocesses may be accepted only when they improve the model as a whole without raising any additional questions.

(iii) The behavior of water quality components indicate some essential differences among lake segments and this is especially valid for the Keszthely Bay when compared to other parts of the lake. Here the eutrophication is in the most developed state and additional factors such as nitrogen limitation, the presence of blue-greeen algae, anaeroby, etc., may play a more pronounced role. Accordingly, several strategies of development may be applied: to concentrate on the accurate modeling of Keszthely Bay, the management of which is the most urgent need and for which the loading history is best defined through the Zala River data, or to find a compromise solution for the whole lake. A combination of these two versions may also be possible. However, from the objective of the eutrophication modeling, namely to describe the water quality of the whole lake, it seems to be unwise to restrict the study to just one part of the lake, the more so since definite interactions exist which make an autonomous development in water quality for separate lake sections unlikely.

(iv) All three models are in the stage of calibration, but have still not been validated. Here the following indispensible steps have to be performed: first to find the parameter values for a certain period and, second, to apply the model with these parameter values for another period (both in the past). If the agreement with the measured data is acceptable the validation can be considered successful, if this is not so, further model modification is needed, guided by the discrepancies observed.

(v) Generally, parameter values are found by trial and error fitting. Here, any better defined method is preferred by all means. The pleasing structure of SIMBAL is again stressed since it permits not only the estimation of the parameters, but also the use of the model then in a stochastic sense, which could be of great importance in the context of management.

Sometimes it is possible to disaggregate the model for the purpose of the calibration. For instance, the parameters of horizontal mass exchange, deposition, and resuspension may be estimated from a total-P model and than fixed in the estimation of remaining parameters in the full lake model. Another example is the separate estimation of phytoplankton growth rates from primary production measurements.

(vi) Finally, the importance of model sensitivity analysis (see also III.4.) and the use of the same input data (such as nutrient loading) for all three models have to be underlined.

3. NUTRIENT LOADING MODELS

The direct cause of artificial eutrophication is the increase in nutrient loading and consequently, this is the most obvious factor for controling the lake. The role of nutrient loading should be emphasized also from the point of view of understanding the in-lake processes and for calibrating LEM (see Figure 7): since it varies both in time and space, neither the dynamic nor the spatial behavior of the lake's water quality can be described until a sound knowledge on the loading conditions is available.

Here, difficulties arise mainly from the lack of data (effluents, sewage discharges, and distributed loadings from the watershed as a whole). Thus it is not easy to make estimates for the total and available loadings, to separate point and non-point sources, and to judge their relative importance, nor is it easy to verify any non-point source model. These kind of models need detailed field data and the problems of how to apply them for a larger watershed or how to perform an extrapolation from a detailed subwatershed study have in general not yet been resolved.

Accordingly our plans for the future are rather modest and the main objective is to make a more thorough loading estimation using existing data rather then concentration on modeling. Here first of all the regular measurements of

Joó (1980) (daily for discharge, temperature, suspended solids, total phosphorus and nitrogen, respectively, weekly for orthophosphate P and complete water chemical analysis) of two stations along the Zala River (Fenékpuszta 1975-8 and Zalaapáti 1977-8, difference in catchment area is roughly 40%) will be used.

The following activities will be performed:

(i) Time series analysis for the Zala River catchment area data. The aim is to develop a simple descriptive model to express the present loadings (first on a daily basis, and second, for a longer time scale, e.g. a week) as a function of parameters that may be easily measured (precipitation, streamflow). By comparing available and total phosphorus data, taking into account the nitrogen data and the existing knowledge about sewage discharges in the watershed, an attempt will be made to separate out that part of the available nutrient loading that is correlated to rainfall events. In this way it is hoped that the role and contribution of agriculture and other non-point sources may be assessed. This is, of course, information of paramount importance in the design of future loading reduction strategies.

In case of success, a next step may be to look for relationships between the various loading types and some of the average socio-economic watershed characteristics. These relationships would then allow the use of the time series model to make predictions under changed conditions for the purpose of planning.

(ii) It is intended to make a parameter sensitivity analysis for the non-point source model explained in Chapter II.4. and utilize the experiences gained for item (i).

(iii) After collecting the existing data for effluents and sewage discharges a similar, but more detailed analysis will be made for the loading a function of time and space as previously made by van Straten et al. (1979) (see also Chapter II.1.). Here the conclusions of (i) and (ii) will have to be incorporated as well.

(iv) For most effluents, generally, not more than one water quality measurement a month is available over a period of 4-5 subsequent years so that any kind of loading estimation (e.g. yearly average or year averaged monthly value) is based on these data. Since more frequent data are given for the Zala River, these can be used for developing methods for calculating the loading from scarce data on the "most accurate" way and for judging the uncertainties of this kind of estimations.

(v) Finally, to judge the importance of changes in watershed activities (e.g. fertilizer use, runoff and erosion control, crop management) a more comprehensive watershed nugrient loading model (such as CREAMS) may be used, but only in a relative sense because of the limited availability and reliability of the data.

4. WATER QUALITY MANAGEMENT MODEL

Since comprehensive management modeling presupposes the availability of well calibrated and validated submodels, which in turn must be based on a sound understanding of the principal processes in lake and watershed (see Figure 7), it should be no surprise to see that management models are in a less progressed stage in the case study. The two approaches mentioned in Chapter II.5. were pursued in the initial phases of the study because they do not rely explicitly on a lake water quality model as the reaction of the lake to nutrient loadings was either inferred from historical data or postulated by an engineering guess. There is, however, little doubt that the use of submodels such as NLM and LEM will be compelling if management models are to yield answers with sufficient credibility. As pointed out in the introduction to this chapter, it is therefore essential to develop methodologies, even in the temporarily absence of sufficiently reliable submodels, in such a way that the results of further research in this area can be gradually incorporated in the management framework without difficulty.

The aim of the management model is to provide a structure that enables the ranking of possible management options according to certain economics, water quality, or mixed criteria (see Figure 8). For the sake of the present discussion, it is convenient to represent Figure 7 and 8 as a circular process as in Figure 15 (see Spofford 1976). In this process the following steps may be distinguished, which can be studied and realized more or less independently before integration:

(i) $L(x,t) \rightarrow$ LEM $\rightarrow I(x,t)$. In this step the response of the lake $[I(x,t)]$ to various loading scenario's is obtained. Initially, reasonable loading scenario's can be postulated, while in a later stage they may be provided by

(ii) loading reduction options $\rightarrow$ NLM $\rightarrow L(x,t)$. This step provides the loadings resulting from various loading reduction options in the watershed together with economic and other aspects;

(iii) The link between (i) and (ii) completes the decision cycle process. Here, the generation and ranking of options is performed in steady feed-back with the other steps. This will involve some kind of (multi-objective) optimization taking costs, benefits, water quality, and socio-economic aspects into account.

Next these steps will be reviewed in more detail.

(i) The method to be followed here is to run the model LEM for different reasonable $L(x,t)$ scenarios and then to parametrize the lake's response in terms of indicators $I(x,t)$ (see III.1.) for use in the optimization procedure. Before doing this, a distinction will be made between forcing functions of the LEM that are natural factors (temperature, radiation, and wind) and those that are controllable factors. The latter is solely the loading $L(x,t)$. Since the variability of the natural factors should be considered as given, the uncertainty resulting from them has to be included. Here, two techniques can be used:

-- to assume different scenarios (average, unfavorable, etc.; dry or wet years, etc.) for studying the sensitivity of $I(x,t)$ to natural factors. This approach will lead to standard scenarios to be used in the subsequent analyses;

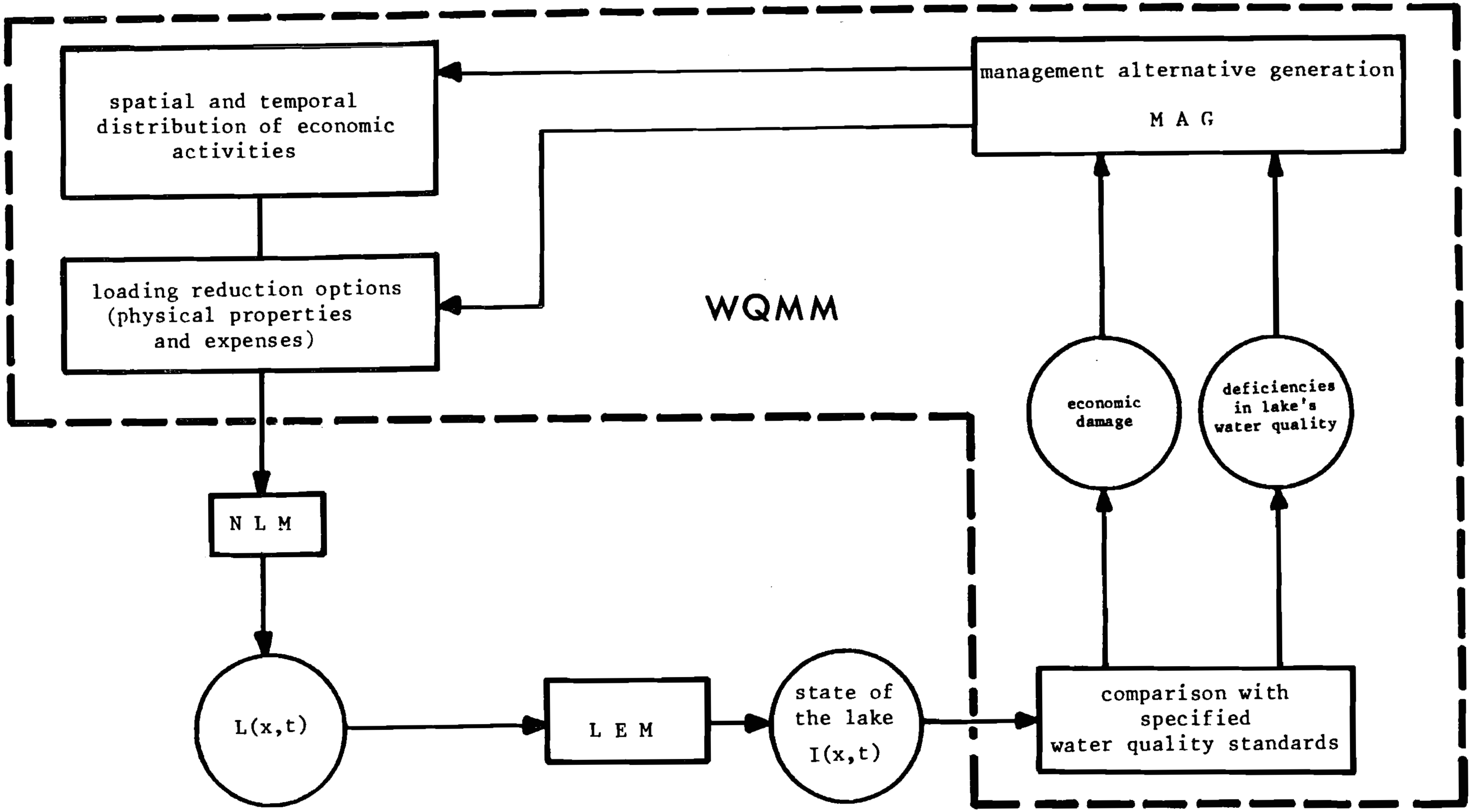


Figure 15. Framework of lake's water quality management.

-- to consider natural factors as basically stochastic in character. The possibilities in this respect will be discussed later.

When designing various loading scenarios for application in LEM, first the uncontrollable fractions (the atmospheric and internal sources) will be separated, thus defining the lowest level for the loading that might be realized. Next, a further classification will be made based on several distinct features such as:

-- availability of nutrients for algae;
-- origin (sewage, agriculture, etc.);
-- location and spatial pattern;
-- temporal changes.

(Also note at this step the economic feasibility of different alternatives has to be considered (Figure 15).)

Having the basic loading types, $L(t,x)$, the lake response $I_i(T,x)$ will be calculated (the subscript refers to the different kinds of indicators needed for characterizing the lake's water quality). Here, T indicates that the latter also changes with time, however, on a larger scale. It is worthy to mention that $I_i(T,x)$ may differ for BEM, BALSECT, and SIMBAL and the level of model simplification in the management context will be governed by the sensitivity of $I_i(T,x)$.

The approach outlined above may only indicate the average and some of the extreme behavior patterns of the system and next, the role of uncertainty will be considered. Various uncertainties result from several factors, such as the stochastic character of all the input forcing functions (see before) and uncertainties in the parameters. Their propagation in the model is of primary interest. For this purpose a non-linear Kalman filtering technique (B. Beck et al. 1979) or a Monte Carlo type of simulation can be applied (note that in the first case the assumption of Gaussian distribution is required). Since the second method is a natural extension of the scenario concept and well suited to the structure of SIMBAL, which has the smallest computer time requirements of the models, its application is planned for the future. Naturally it will not be realistic to perform the study of error propagation for all the cases considered and therefore some typical examples will be selected in the hope that these results can be generalized.

(ii) From inspection of the different loading sources it is a simple matter to make a list of possible reduction alternatives. Among the great number of alternatives here only sewage treatment (mainly tertiary) and the application of reed-lakes will be reviewed, since these are felt to be presumably the most effective tools for removing available nutrient forms. The former is applied before discharging waste water to the lake or rivers of the catchment, while the latter should be located on those rivers (before entering the lake, which are the recipients of several sewage discharges.

A reed-lake system removes nutrient by sedimentation (of particulate forms) and benthic eutrophication (of available forms). In addition such a system has a considerable storage capacity for equalizing sudden changes in the loading caused by rainfall events and therefore it could be effective for reducing the time variable loading originating from agriculture. It should be noted here that although the realization of the first stage of such a system on the Zala River will start next year (Kis-Balaton, see van Straten et al. 1979) many open questions have to be answered in the immediate future, both related to planning (e.g. what is the relation between design and removal efficiency, what are the investment and operational costs, etc.) as well as operation (e.g. throughflow control for optimal efficiency, when to harvest reeds, how to utilize the reeds, ect.). It is also mentioned that due to the seasonal pattern of benthic eutrophication the reed system is less capable of removing dissolved nutrients in winter. Furthermore, organic loading must be at a level low enough to prevent anaerobic conditions in the system.

In the light of just these two options the need for surveys of existing and planned sewage plants, of the possibilities for addion tertiary treatment to operating plants, of canalization, and of the feasibility of establishing reed-lakes and sludge deposition systems is apparent. Location, timing, capacity, efficiency and expenses have to be included in the study. This then as a whole may lead to the development of a regional concept for reducing nutrient loading. Naturally, other control activities also have to be considered and this kind of background work is being undertaken at VITUKI and SZTAKI.

(iii) When all these steps are completed WQMM will be carried out as shown in simplified form in Figures 8 and 15. Here, it is merely stated that WQMM should yield a few "not bad" combinations of different control alternatives not only in a global sense but as a (discrete) function of both space and time. With this result the decision makers should be able to select the "best" solution on the basis of their preferences, or to initiate a new planning cycle if the result was not satisfactory.

APPENDIX A

Hungarian Collaborative Institutes - Abbreviations

MTA	Hungarian Academy of Sciences
MAREB	Hungarian Committee for Applied Systems Analysis
MTA SZTAKI	Computer and Automation Institute of the Hungarian Academy of Sciences
MTA BKI	Biological Research Institute of the Hungarian Academy of Sciences, Tihany
OVH	National Water Authority
VITUKI	Research Centre for Water Resources Development

Members of the Subcommittee of the Hungarian Committee for Applied Systems Analysis responsible for the Balaton Case Study Cooperation:

Chairman	:	Dr. P. Benedek (VITUKI)
Secretary	:	Dr. P. Literáthy (VITUKI)
Members	:	Prof. A. Prékopa (MTA SZTAKI)
		Prof. Juhász-Nagy (University of Budapest)
		Dr. L. Dávid (OVH)
		Dr. B. Entz (MTA BKI)

APPENDIX B

Visits of Hungarian Scientists to IIASA: Frame of Lake Balaton Case Study 1978-1979

1978

June 12-25	I. Bogárdi (Mining Research Institute) Stochastic Input Modeling (with L. Duckstein)
June 26-30	I. Wittman (MTA SZTAKI) P loading model
July 10-14	J. Fischer (MTA SZTAKI) Organizational aspects
July 10-14	A. Szöllosi-Nagy (VITUKI) Parameter estimation
July 13-21	Gy. Hoffmann (MTA SZTAKI) Sediment model
August 21 - September 1	P. Csáki (MTA SZTAKI) L. Hajdú (Natural Historic Museum) S. Herodek (MTA BKI) G. Jolánkai (VITUKI) BEM Model activities
November 20-24	L. Dávid (OVH) Watershed Development approach

1979

March 4-7	L. Somlyódy (VITUKI)	
		Hydrodynamic Models
March 26- April 6	G. Jolánkai (VITUKI)	Problem Analysis report
April 2-6	L. Dávid (OVH) L. Telegdi (MTA SZTAKI)	
		Watershed development approach
May 28 - June 1	I. Wittman (MTA SZTAKI)	
May 28 - June 2	L. Somlyódy (VITUKI)	Hydrodynamics
May 23 - June 8	T. Kutas (MTA SZTAKI) S. Herodek (MTA BKI)	
		BEM Model
June 7-21	I. Bogárdi (Mining Research Institute)	
June 11-19	L. Dávid (OVH)	
		Multicriteria decision model(with L. Duckstein)
July 9-13	G. Jolánkai (VITUKI)	
July 9-20	A. Szöllösi-Nagy (VITUKI)	
		Parameter estimation
July 9-20	T. Kutas (MTA SZTAKI) S. Herodek (MTA BKI)	
		BEM model
July 16-27	J. Fischer (MTA SZTAKI)	
		Organizational aspects
October 1-5	A. Prékopa (MTA SZTAKI)	
		Operations Research orientation
October 15-26	L. Somlyódy (VITUKI)	
		Hydrodynamics
November 26-30	E. Dobolyi (VITUKI) T. Kutas (MTA SZTAKI) J. Fischer (MTA SZTAKI) S. Herodek (MTA BKI)	
		BEM modeling discussion
November 26-30	L. Kovács (MTA SZTAKI) A. Prékopa (MTA SZTAKI)	
		Operations Research aspects

REFERENCES

Beck, M.B. 1978. Mathematical Modeling of Water Quality. CP-78-10. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Beck, M.B., E. Halfon, and G. van Straten. 1979. The Propagation of Errors and Uncertainty in Forecasting Water Quality--Part I. Method. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Bogárdi, I. and L. Duckstein. RM-79-033. A Stochastic Model of Phosphorus Loading from Non-point Sources. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Boyce, F.M, A.S. Fraser, E. Halfon, D. Hyde, D.C.L. Lam, W.M. Schertzer, A.H. El-Shaarawi, T.J. Simons, and K. Willson of the Applied Research Divsion and D. Warry of the Water Quality Branch, project leader—T.J. Simons. Assessment of Water Quality Simulation Capability for Lake Ontario. Scientific Series No. 111. Inland Water Directorate, National Water Research Institute, Canada Center for Inland Waters: Burlington, Ontario.

Csáki, P. and T. Kutas. 1980. The BEM Modeling Approach: Development of a Model for Lake Balaton's Ecosystem. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fischer, and I. Kovacs—Editors. MTA: Veszprém, Hungary.

Dávid, L., L. Telegdi, and G. van Straten. CP-79-016. A Watershed Development Approach to the Eutrophication Problem of Lake Balaton (A Multiregional and Multicriteria Model). Laxenburg, Austria: Internaional Insitute for Applied Systems Analysis.

Dobolyi, E. 1979. Identification of Different Phosphorus Forms and their Role in the Eutrophication Process of Lake Balaton. Presented at the SIL Workshop on Hypertropic Ecosystems, October: Växjö, Sweden.

Dobolyi, E. and S. Herodek. 1980. On the Mechanism Reducing Phosphate Concentration in the Water of Lake Balaton. Budapest: Internationale Revue de. gesamten Hydrobiologie, 65.3.1980: 339-343.

Duckstein, L., I. Bogárdi, and L. Dávid. 1980. Multi-objective Control of Nutrient Loading into a Lake. IFAC Symposium on Water and Related Land Resource, May, 1980, Cleveland, Ohio.

Durham, L.D. and L.H. Butler. 1976. Lake Erie International Jetport Model Feasibility Investigation. Miscellaneous Paper, H-76-3. Vicksburg, Mississippi: U.S. Army Engineer Waterways Experiment Station.

Fisher, I. 1980. Representation of Hydrophysical and Ecological Processes in the Simulation of Water Quality in Shallow Lakes. A publication of the resouce Engineering Department of the Univeristy of New England, Armidale, New South Wales— Australia.

Fisher, I. 1980. International Institute for Applied Systems Analysis. Memorandum within the Resources and Environment Area, February 20, 1980.

Fisher, I. 1980. Procedure for Obtaining Wind-induced Circulation Patterns for a Lake Using the Laplace Transform Model of Young and Liggett (1977). International Institute for Applied Systems Analysis. Internal Report, January, 1980.

Gedney, R.T. 1971. Numerical Calculations of the Wind-driven Currents in Lake Erie. Technical Memorandum TMX-52985. Cleveland, Ohio: U.S. National Aeronautics and Space Administration.

Gedney, R.T. and W. Lick. 1972. Wind-driven Currents in Lake Erie. Journal of Geophysical Research, Vol. 77 (15): 2714-2723.

Györke, O. 1975. Water and Sediment Motion in the South-Eastern Area of Lake Balaton Characterizing Bed Formation (in Hungarian). Budapest, Hungary: VITUKI report.

Halfon, E. and D.C.L. Lam. 1978. The Effects of Advection-Diffusion Processes on the Eutrophication of Large Lakes, A Hypothetical Example: Lake Superior. Ecological Modelling 4 (1978), Elsevier Scientific Publishing Company, Amterdam, The Netherlands: 119-131.

Harleman, D.R.F.H. and P. Shanahan. 1980. Aspects of Wind-driven Circulation and Mixing in Eutrophication Studies of Lake Balaton. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fischer, and I. Kovács—Editors. MTA: Veszprém, Hungary.

Herodek, S. 1977. Recent Results of Phytoplankton Research in Lake Balaton. Annales Instituti Biologici (Tihany) Hungaricae Academiae Scientiarum, Vol. 44: 185-189.

Herodek, S. and P. Csáki. 1980. The BEM Modeling Approach: Ecological Aspects of Eutrophication Modeling. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fischer, and I. Kovács—Editors. MTA: Veszprém, Hungary.

Hirling, Gy. 1979. Meteorological Conditions in the Region of Lake Balaton. In a summary of research results for Lake Balaton, edited by S. Baranyi, in Hungarian. VIZDOK: Budapest.

Jolánkai, G. 1977. Field Experiments and Modeling of Non-point Source Pollution on a Subwatershed of Lake Balaton. Proceedings of the 17th Congress of the International Association of Hydraulic Research, August 15-24, 1977, Baden-Baden, Austria.

Jolánkai, G. 1980. Plant Nutrient Balance Considerations for Lake Balaton. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fishcer, and I. Kovács—Editors. MTA: Veszprém, Hungary.

Joó, O. 1980. The Role of the Zala River in the Eutrophication of the Balaton. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fischer, and I. Kovács—Editors. MTA: Veszprém, Hungary.

Jørgensen, S.E. and D.R.F. Harleman. 1978. CP-78-7. Hydrophysical and Ecological Models of Deep Lakes and Reservoirs. Laxenburg, Austria: International Insitute for Applied Systems Analysis.

Jørgensen, S.E. 1979. CP-78-14. Hydrophysical and Ecological Models of Shallow Lakes and Reservoirs. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Kutas, T. and S. Herodek. 1980. The BEM Modeling Approach: Recent Results in Simulating Lake Balaton's Ecosystem. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. G. van Straten, S. Herodek, J. Fischer, and I. Kovács—Editors. MTA: Veszprém, Hungary.

Kvon, V.I. 1977. Mathematical Simulation of One-dimensional Thermocline. Paper presented at an IIASA Workshop on Geophysical and Ecological Modeling of Deep Lakes and Reservoirs, December 12-15, 1977.

Lam, D.C.L. and E. Halfon. 1978. Model of Primary Production, Including Circulation Influences, in Lake Superior. Reprinted: Applied Mathematical Modelling by Editor-C.A. Brebbia, IPC Science and Technology Press, Southhampton, England: 1978, Vol. 2, March.

Launder, B.E. and D.B. Spaulding. 1972. Lectures in Mathematical Models of Turbulence. London: Academic Press.

Leonov, A.V. 1980. WP-80-88. Simulation and Analysis of Phosphorus Transformation and Phytoplankton Dynamics in Relation to the Eutrophication of Lake Balaton.

Muszkalay, L. 1979. Water Motions in Lake Balaton. In a summary of research results for Lake Balaton. Edited by S. Baranyi, in Hungarian. VIZDOK: Budapest.

Oláh, J., L. Tóth, and E. Tóth. 1977. Phosphorus Household of Lake Balaton. MTA Biol. Oszt. Közl., No. 20, (In Hungarian): 111-139.

Somlyódy, L. 1979. CP-79-1. Hydrodynamical Aspects of the Eutrophication Modeling in the Case of Lake Balaton. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Somlyódy, L. 1980. Preliminary Study on Wind Induced Interaction between Water and Sediment for Lake Balaton (Szemes Basin). G. van Straten, J. Fischer, and I. Kovács—Editors, Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. MTA: Veszprém, Hungary.

Spofford, W.O., Jr. 1976. Integrated Residuals Management: A Regional Environmental Quality Management Model. (Originally titled "Total Environment Quality Management Models" -- reprinted from "Models for Environmental Pollution Control" by Ann Arbor Science Publishers, Inc.), Resources for the Future, Reprint 130, Washington, D.C.

Van Straten, G., G. Jolánkai, and S. Herodek. 1979. CP-79-13. Review and Evaluation of Research on the Eutrophicatio of Lake Balaton - A Background Report for Modeling. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Van Straten, G., S. Herodek, J. Fischer, and I. Kovács—Editors. 1980. Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. MTA: Veszprém, Hungary.

Van Straten, G. 1980. WP-80-139. Analysis of Model and Parameter Uncertainty in Simple Phytoplankton Models for Lake Balaton. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Watanabe, M., D.R.F. Harleman, and O.F. Vasiliev. 1980. Two- and Three-Dimentional Mathematical Models for Lakes and Reservoirs. Chapter 8 in Mathematical Modelling of Water Quality: Streams, Lakes, and Reservoirs, G.T. Orlob, editor. Forthcoming.

Young, F.D.L. and J.A. Liggett. 1977. Transient Finite Element Shallow Lake Circulation. Journal of the Hydraulics Division, Vol. 103, No. HY1, February 1977, American Society of Civil Engineers: 109-121.

IIASA PUBLICATIONS RELATED TO THE LAKE BALATON CASE STUDY

1978

RM-79-033. Bogårdi, I. and L. Duckstein. A Stochastic Model of Phosphorus Loading from Non-point Sources.

1979

Proceedings of the Second Joint MTA/IIASA Task Force Meeting on Lake Balaton Modeling. van Straten, G., S. Herodek, J. Fischer, and I. Kovåcs, editors: Veszprem, Hungary.

CP-79-001. L. Somlyódy. Hydrodynamical Aspects of Long-Term Water Resources Planning in the Case of Lake Balaton.

CP-79-013. van Staten, G., G. Jolânkai, and S. Herodek. Review and Evaluation of Research on the Eutrophication of Lake Balàton--A Background Report for Modeling.

CP-79-016. Dåvid, L., L. Telegdi, and G. van Straten. A Watershed Development Approach to the Eutrophication Problem of Lake Balaton (A Multiregional and Multicriteria Model).

1980

WP-80-88. A.V. Leonov. Simulation and Analysis of Phosphorus Transformation and Phytoplankton Dynamics in Relation to the Eutrophication of Lake Balaton.

WP-80-139. G. van Straten. Analysis of Model and Parameter Uncertainty in Simple Phytoplankton Models for Lake Balaton.

WP-80-149. A.V. Leonov. Mathematical Modeling of Phosphorus Transformation in the Lake Balaton Ecosystem.

Forthcoming Publications

WP. Somlyódy, L. and M. Virtanen. Application of a One-dimensional Hydrodynamical Model to Lake Balaton.

WP. Shanahan, P., Harleman, D.R.F., and L. Somlyódy.
Modeling Wind-driven Circulation in Lake Balaton.

WP. Somlyódy, L. and J. Eloranta.
Use of a Lake Eutrophication Model for
Management Purposes.

CP. van Straten, G. and S. Herodek.
Estimation of Algal Growth Parameters from
Vertical Primary Production Profiles.

RR. Bogárdi, I., L. Dávid, and L. Duckstein.
Multicriterion Control of Nutrient Loading Into
a Water Body.