EUTROPHICATION OF LAKES AND RESERVOIRS IN WARM CLIMATES

Lars Landner
Ulf Wahlgren

Swedish Environmental Research Group
Stockholm, Sweden

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NOTE

FOREWORD

Within the framework of the Health for All by the Year 2000 policy, the European strategy is consolidated into 38 targets. Target 20 is related to the protection of the quality of water resources used to produce drinking water.

Besides primary pollution from discharge of liquid waste into natural waters, lakes and reservoirs may suffer from secondary pollution from hydrobiological eutrophication. Limnological studies of lake eutrophication first started in the European countries, such as Sweden and Switzerland, that are most dependent upon lakes as water sources. Therefore, the scientific knowledge on hydrobiological processes in lakes and reservoirs, including eutrophication, has mainly been developed in temperate/cold climates. Under these climatic conditions, the temperature of the full body of water becomes uniformly less than 4>C twice a year, with a resulting turn-over of the water layers.

Some countries, such as Sweden and Switzerland, have enough natural lakes to provide drinking-water supplies without the need to build reservoirs. However, in Mediterranean countries, where natural lakes are few and small, the need to store winter rainfall has led to the building of numerous dams to create water reservoirs.

In the Mediterranean area, the hydrobiological processes of reservoirs, including eutrophication, are quite different from those found in natural lakes in cold climates. Unlike natural lakes, reservoirs are young bodies of water that have not achieved their ecological balance. In addition, the deep-water layer, called the hypolimnion, keeps a constant temperature above 4>C, therby preventing the biannual turn-over of water layers characteristic of temperate/cold climates.

The very few well-documented studies available on the eutrophication processes under temperate/warm climates have been conducted in California in the USA and in Australia. The WHO-assisted project in Morocco has provided the opportunity to develop a large-scale study with the collaboration of scientists from the Swedish Environmental Research Group (SERG). The results of this study are consolidated in this volume together with the results of other studies in warm-climate countries. It is hoped that this volume will be a valuable contribution to the global scientific knowledge of hydrobiology and protection of the quality of water reservoirs.

I wish to express my appreciation to Drs Lars Landner and Ulf Wahlgren for preparing this volume.

Eric Giroult
Regional Officer for Environmental
Health Planning/Ecology

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PART 2: CASE STUDIES

PART 1

EUTROPHICATION: PROBLEM AND CONTROL

1. INTRODUCTION

A large number of studies on the process of eutrophication, its causes, consequences and possible corrective measures, have been conducted during the last two decades. These studies have sometimes been carried out on the national scale but more often as international collaborations because of the complexity of the problem and the multiple negative effects on the use of the water resources for productive purposes. Among the most important international cooperation programmes, the following are particularly noteworthy:

- the first fundamental study of the phenomenon, initiated by the Organisation for Economic Co-operation and Development (OECD) during the 1960s, which gave rise to the famous Vollenweider report [1];

- the interNordic programme which was conducted during the first part of the 1970s [2];

- the cooperative programme conducted by OECD during the 1970s which resulted in four reports on regional eutrophication studies (mostly concerned with lakes and reservoirs in cold and temperate climates in Europe and North America) and in a comprehensive report on the whole cooperative programme [3];

- the preparation of a global document on eutrophication control in lakes and reservoirs by UNESCO in the early 1980s [4].

In the early 1970s, the World Health Organization (WHO) became interested in the eutrophication problem. At this time WHO formed an international working group on eutrophication and initiated a comprehensive study on lake eutrophication. This working group resulted in a report on causes, effects and control measures with an emphasis on the restoration of already eutrophied lakes [5].

All the efforts hitherto mentioned have emphasized lakes and reservoirs in temperate or cold climatic regions. When WHO, together with the United Nations Development Programme (UNDP), initiated a study of the eutrophication of the drinking-water reservoir Sidi Mohammed Ben Abdellah (SMBA) in Morocco in 1978, the need to examine the possible differences between eutrophication processes under different climatic conditions became imminent.

Several round-table discussions and international seminars were organized at the completion of the SMBA project in 1982. The final report from the SMBA project also served as a basis for other eutrophication studies in the three countries of the Maghreb: Algeria, Tunisia and Morocco.

During the early 1980s, several national and international programmes on eutrophication were executed. One project that in particular deserves to be mentioned is a regional programme for developing simplified methods for evaluating the state of eutrophication in warm lakes in Latin America. This project was coordinated by the Pan American Centre of Sanitary Engineers and Environmental Sciences (CEPIS) of the Pan American Health Organization (PAHO). The achievements obtained in this project may be found in Castagnio [6] and Salas [7].

In 1984, the WHO Regional Office for Europe decided to document the experiences to date on eutrophication of lakes and reservoirs in warm climates. The SMBA study served as the basis for this work, complemented with data from studies in other subtropical or tropical regions.

The present document consists of two parts. Part 1 presents the problem and discusses the causes, effects and methods used in eutrophication studies and summarizes possible measures to counteract the negative effects of eutrophication of lakes and reservoirs. Part 2 consists of two concrete cases taken from the WHO/UNDP eutrophication studies in North Africa.

This volume is not a general manual on eutrophication studies in warm climates. Rather, its aim is to discuss some important aspects of the peculiarities of eutrophication in warm climates based on, among other things, the North African experiences and to highlight the differences between the eutrophication process in warm and temperate climates.

2. THE PROCESS OF EUTROPHICATION AND ITS REPERCUSSIONS

2.1 Definition

During the last two decades, the term eutrophication has frequently been used to describe the artificial and involuntary enrichment of a water body by nutritional elements, in particular nitrate and phosphate. This use of the term may give rise to a certain confusion, as a given quantity of nutrients transported to one water body may produce undesirable effects while the same load may not cause any noticeable damage or even sometimes produce positive effects in another lake or reservoir.

The term eutrophication is, however, normally used to describe a phenomenon of excessive algal growth or growth of higher aquatic plants as a result of an excessive supply of nutrients. An enhanced algal growth followed by an accumulation of algal biomass and of detritus in the water mass will in general bring about a degradation of the water quality, which interferes significantly with normal multiple uses of the resource. Vollenweider [1] has defined eutrophication as

> an enrichment of nutrients and the progressive deterioration of the water quality, in particular for lakes, due to an excessive algal growth with all its effects on the metabolism of the affected water.

More recently, OECD has defined eutrophication as

> the enrichment of nutrients in a water body which causes a stimulation of a large number of symptomatic changes among which are an augmentation of the production of algae and macrophytes, the deterioration of the water quality and other changes which are negative and interfere with the usage of the affected waters.

These definitions of eutrophication evolved because eutrophication has come to be viewed as a water quality problem during the last decades. Nevertheless, the term eutrophication in its most original sense means a natural aging of a lake. With time, a lake progressively fills with particulate matter transported by the rivers entering the lake. The lake thus slowly but surely transforms into a swamp and finally into a terrestrial system. This process usually takes thousands of years, and it is in principle irreversible. Lakes subjected to natural eutrophication usually have a good water quality and are characterized by highly diverse biological communities. In lakes with "virgin" basins, that is, where humans have not intervened, the production of algae and other aquatic plants is often in equilibrium with the supply of nutrients.

2.2 Man-made eutrophication

As soon as human activities, such as forestation, agriculture, and construction of villages and cities, intervene in a lake basin, the load of nutrients usually increases because of increased erosion, chemical mobilization of elements normally bound in the surface layer of the soil, overfertilization and direct discharges from human settlements.

One may consider a lake or a reservoir as a trap for all material transported by the water entering the lake. Thus, the lake can be expected to accommodate the integrated effect of all the modifications of the natural and the additional purely man-made processes that have taken place in the basin. If these changes have brought about increases in the nutrient load, the lake's response is almost always a stimulated growth of algae and other aquatic plants and, frequently, a stimulated production at higher trophic levels in the system (e.g. zooplankton, fish). To distinguish this phenomenon from natural eutrophication processes, the term "cultural eutrophication" or "artificial eutrophication" is used. Cultural eutrophication is a fast process in comparison with its natural counterpart, and it is reversible to a large extent, although this may demand substantial efforts. As cultural eutrophication entails negative environmental effects and is caused by human activities, it can logically be considered in the general context of water pollution.

When the eutrophication problem is considered, several aspects of the problem must be distinguished:

- eutrophication in its most narrow sense (i.e. an augmentation of the load of nutrients);

- the biological response and the factors that may modify the response;

- the more or less visible or easily detectable symptoms of eutrophication;

- the consequences of eutrophication for the uses of the water resource, including negative effects on public health and on higher organisms in the trophic system.

The biological response of a lake or reservoir exposed to an increased nutrient load is not necessarily a luxurious algal growth. A large number of internal and external factors influence the response of the biological system. Some important external factors are the morphology of the lake (the surface/-depth relationship), the hydraulic situation (periods of drought and flood), the isolation factor and the optical conditions (turbidity and colour of the water). Among the internal factors are the structure of the trophic chain (primary and secondary consumers) and the chemical composition of the water (e.g. the presence of large concentrations of calcium ions).

Apart from the nutrients, many factors determine the productivity of a lake or a reservoir. Important properties of the lake, the incoming water and the limmological system are schematically shown in Figure 1. All properties determining the evolution of the lake are grouped in a hierarchical order where the highest level includes climatological and hydrological factors. These factors determine the properties of the water in the basin independent or almost independent of the presence of a lake [8]. Clearly, the quality of the water reaching a lake does not depend on the lake; it depends only on the properties of the basin and climatological conditions.

These facts bring us to study the differences in the sensitivity to cultural eutrophication that may exist between lakes and reservoirs located in warm climates and those located in temperate regions. To this end, an understanding of the differences between the processes involved in the eutrophication under different climatological conditions is desirable; these questions are discussed in more detail in section 5 of Part 1. In brief, lakes in warm climates show signs of eutrophication more often than lakes in temperate regions because the nutrient loadings often are more important, temperatures are higher and the insolation is more important during longer periods of the year than in the warm regions.

The typical symptoms of cultural eutrophication are generally well known: "flowering" in the water, luxuriant growth of higher aquatic plants, formation of algal foam or carpets of floating algae, bad odour from decomposing algae or dead plants, decreased water transparency, deoxygenation of the deep-water layers followed by production of hydrogen sulfide (a malodorous and highly toxic gas) and massive fish death. These symptoms, which may be more or less serious depending on the degree of eutrophication, are generally easy to observe and become particularly apparent when the water is used for recreational purposes. However, a direct relation does not always exist between the visible symptoms of eutrophication and the problems associated with the water uses (except when the resource is used for recreational purposes).

More and more surface water tends to be used for different purposes, such as raw water for industry, drinking water, and water for irrigation and aquaculture. This increased exploitation of surface water is particularly evident in arid or semi-arid regions, where a large number of reservoirs have been or are being built on rivers. Different water uses necessitate different and specific water quality criteria. If the quality of the raw water does not correspond to the criteria defined for the water use in question, the water must be submitted to a more

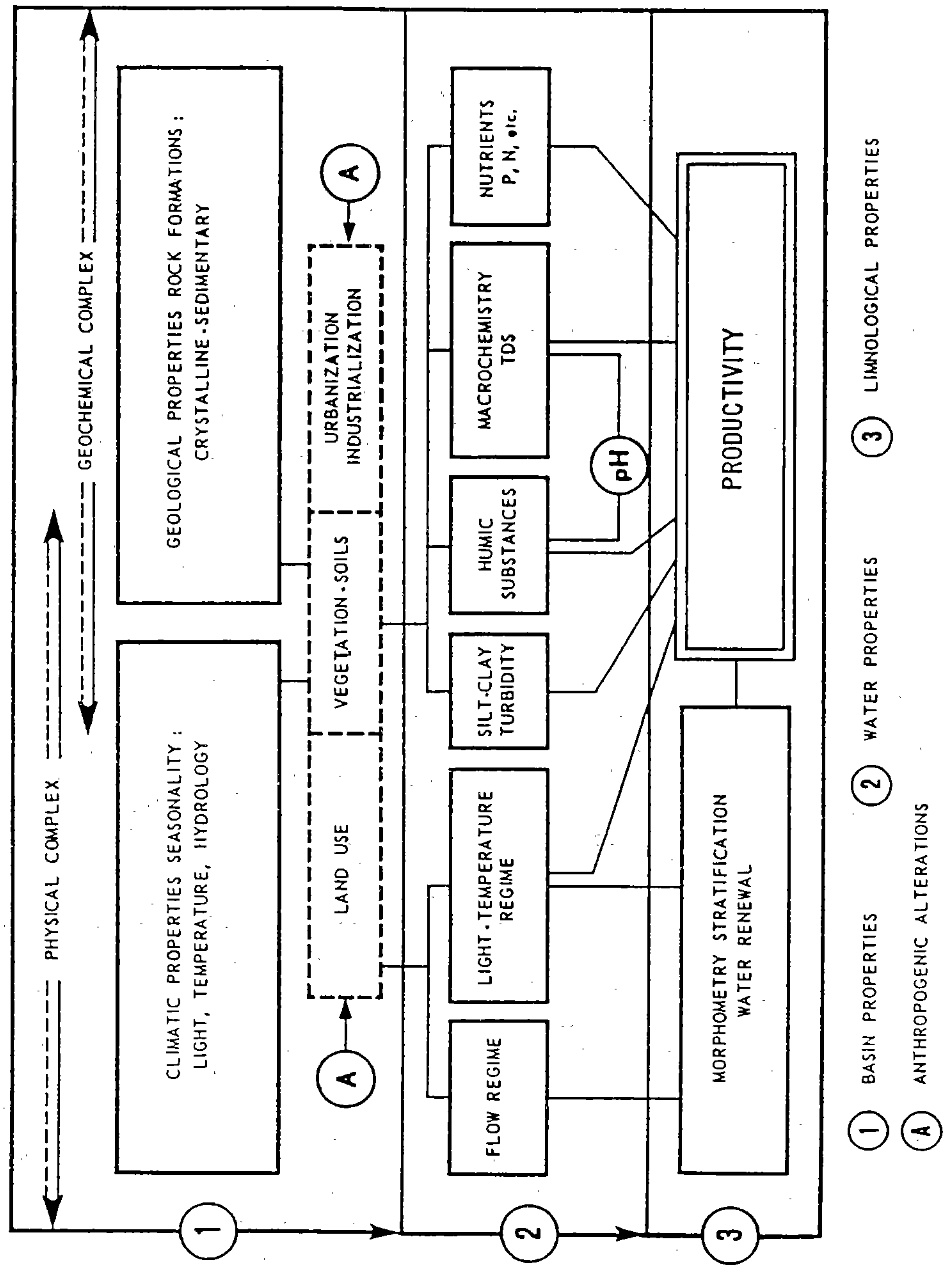


Fig. 1. *The three levels determining productivity of bodies of water [8]*

or less complicated (i.e. costly) treatment. Raw water withdrawn from a eutrophied reservoir is generally of poor quality and must consequently be subjected to an advanced and costly treatment (e.g. by active carbon), which may increase the cost of the water for the consumer to prohibitive levels.

One of the most annoying problems that may occur when a deep eutrophied reservoir is used as a raw water resource for industrial or drinking water is hydrogen sulfide formation in the deep water layers (the hypolimnion). Water withdrawn from the deep layers may be highly contaminated by hydrogen sulfide for a large part of the year. Eliminating hydrogen sulfide from the raw water is very costly, and eliminating bad taste and odour from the water originating from the treatment process in the presence of hydrogen sulfide is very difficult. Independent of the hydrogen sulfide problems, such water also may be less palatable because the anoxic conditions in the hypolimnion may cause high concentrations of certain ions (by release from the sediments), particularly iron, manganese, nitrite and ammonium.

A highly eutrophied reservoir presents several problems other than the presence of hydrogen sulfide (if the reservoir is deep) if it is to be used as a drinking-water source. These more direct consequences are caused by the excessive algal growth and are typically connected with the productive layers of the reservoir, that is, the epilimnion:

- a rise of the pH of the water, which interferes with the flocculation by aluminum sulfate, increasing the need for reactants;

- high concentrations of suspended organic matter, which make the water treatment more difficult and more expensive;

- high concentrations of dissolved organic matter, which increase the quantities of trihalomethanes in the water after chlorination;

- bad taste and odour due to partly decomposed algae;

- appearance of filamentous algae, which have a tendency to clog the filters;

- an increased number of bacteria;

- appearance of parasites in the reservoir.

2.3 Adverse health effects

Few examples of direct health effects on humans resulting from eutrophication are found in the literature. Among the reported effects are the appearance of dermatitis or allergies in individuals who have had contact with water highly contaminated by blue-green toxic algae (certain Cyanophyta), for example, while swimming in a lake. Furthermore, small quantities of such water swallowed during bathing may cause gastroenteritis [9]. A less direct but sometimes very serious effect is the creation of a favourable environment for mosquitoes and molluscs that are vectors for malaria and schistosomiasis, respectively. Studies have shown that eutrophication of lagoons and small lakes has favoured the production of the mosquito genus *Culex*, a vector for several types of viral encephalitis.

Other indirect effects of eutrophication of drinking-water reservoirs should be considered. The concentration of dissolved organic matter in the raw water may attain very high levels in eutrophied reservoirs. The dissolved organic matter is the subject of chlorination in the water treatment. During this process, certain organic compounds, such as chloroform and other trihalomethanes, may be formed in measurable quantities, and as they are reasonably persistent they appear in the tap water. These compounds have been shown to be weak carcinogens. Furthermore, the presence of such compounds serves as an indicator of several other micropollutants whose carcinogenicity is much more severe (see [10]).

A last example of indirect health hazards connected with eutrophication of drinking-water reservoirs is the reaction of the public to tap water with bad taste and odour. Although this water may be perfectly safe to consume, consumers do not accept it and start to use water from other nontreated sources - which may be contaminated with highly toxic or infectious microorganisms.

Eutrophication has other negative consequences that are not health hazards but that may become quite costly to correct. High concentrations of carbon dioxide may corrode pipes and other structures when the water is used for industrial purposes (e.g. for use in hydroelectric plants); these problems become even more accentuated if the water contains high concentrations of hydrogen sulfide. Eutrophied water containing certain toxic blue-green algae (Cyanophyta) may provoke illness and even death among cattle. The dominant algae during toxic water flowering belong to the genera *Microcystis*, *Anabena*, *Aphanizomenon*, *Nodularia* and *Oscillatoria*. Species in these genera all benefit from increased concentrations of nutrients in the water and elevated temperatures.

2.4 Beneficial effects

However, eutrophication also can have positive effects: for example, the development of the fish population in certain cases or use of the water for aquaculture. In these cases the increased aquatic productivity may be largely beneficial, under the condition that it can be contained within well-controlled limits.

3. EUTROPHICATION OF LAKES IN TEMPERATE CLIMATES

3.1 General aspects

As has already been mentioned, most of the work in eutrophication has been carried out on lakes in the temperate zone (i.e. Europe and North America). The experiences obtained through these studies have largely provided the basis for model development and programmes for eutrophication control.

A large body of literature concerns eutrophication of lakes in the temperate and cold zones, and many excellent review articles also are available [1,3-5,11-14].

Although the body of scientific literature on eutrophication in warm climates is growing rapidly, relatively few review articles have been written. The following reports cover most of what has been done in this field: Castagnino [6], Landner [15], Vollenweider [8], Australian Water Resources Council [16], Salas [7] and Thornton [17]. To provide the basis for a thorough discussion on eutrophication in lakes and reservoirs in warm climates, a short presentation of the essential results and conclusions obtained in the studies in the temperate zone is pertinent. The reader who desires a more detailed description of this vast field is referred to the literature quoted above.

3.2 The concept of a limiting nutrient

The growth of algae and other aquatic plants depends on the availability of a number of essential elements, some of them in trace quantities, and on organic growth factors. The chemical macroconstituents necessary for growth of most aquatic plants, including algae, are C, O, H, N, S, P, K, Mg, Ca, Fe and, in certain cases, Si. The most important micro-elements are Mn, Co, Mo, Cu and Zn. The autotrophic growth of plants can be described by a well-known equation describing photosynthesis reaction:

$$6CO_2 + 6H_2O + \text{light} \longrightarrow C_6H_{12}O_6 + 6O_2$$

Taking the chemical composition of the algae into account, the photosynthetic equation may be modified as follows:

$$106CO_2 + 15NO_3^- + HPO_4^{2-} + SO_4^{2-} + 19H^+ + 79H_2O \longrightarrow C_{106}H_{178}O_{50}N_{15}PS + 147O_2$$

The second equation describes the approximate relation between the most important elements in the algal biomass: C:N:P = 106:15:1. It also permits the amount of oxygen produced during the photosynthesis or consumed during degradation of the biomass to be estimated. To accomplish a complete mineralization of 1 g of organic matter originating from algae, about 1.8 g of oxygen is consumed. If the organic matter is measured in units of carbon, about 3.6 g of oxygen is needed to transform 1 g of organic carbon into CO_2.

The concept of a limiting nutrient signifies the element existing in the weakest concentrations in relation to the limiting (or determining) need for the plant growth. In reality, this concept, based on "the Law of Liebig", is a gross simplification because the chemical form, the absolute concentrations, and the interaction with other nutritional elements and the plant cellules play an important role in determining the final biological response. Nevertheless, the concept of a limiting nutrient has proven to be very useful for the describing and understanding the eutrophication process and, not least important, for designing effective remedial measures. It should be underlined that any of the elements mentioned above can, for shorter or longer time periods, be the limiting factor for growth. In practice, however, only a few of these elements become limiting for any significant period of time: in particular nitrogen and phosphorus and, to a lesser extent, carbon and/or certain micro-elements. In certain cases light may also be the limiting factor. In the majority of cases, however, the factor controlling the production is phosphorus [8].

3.3 Indicators of trophic state

When lakes and reservoirs were first classified according to their nutritional status, the categories used were oligotrophic (from the Greek term "little nourishment") and eutrophic ("well fed"). These terms originally described the fertility of peat bogs in northern Germany [18]. Later, these concepts were transferred to lakes by Nauman [19] and Thienemann [20]. The term mesotrophic was then introduced to describe lakes in transition between the oligotrophic and eutrophic states. During recent decades, the terminology has been further expanded with the terms ultra-oligotrophic for extremely nutrient-poor systems and hypertrophic for extremely productive waters, such as fish production ponds.

The classification of lakes into different trophic categories is largely qualitative and should be carried out only by experienced observers. The meaning of eutrophic and oligotrophic may even differ for different individuals or in different limnological situations; observers used to oligotrophic lakes are probably more sensitive to signs of degradation than those from regions characterized by more productive lakes. General characteristics of oligotrophic and eutrophic lakes have been compiled in the UNESCO report [4]: Table 1 gives a summary.

Table 1. *General characteristics of oligotrophic and eutrophic water bodies in the temperate zone*[1]

Parameter	Oligotrophic	Eutrophic
I. Biological		
Production of plants and aquatic animals	Low	High
Number of plant and animal species	High	High, but reduced in hypertrophic waters
Biomass	Low	High
Occurrence of algal flowerings	Rare	Frequent
Frequency of green or blue-green algae	Low	High
Distribution of algae	In epilimnion and hypolimnion	Only in epilimnion

Table 1. *(cont'd)*

Aquatic plants in littoral zone	Rare or abundant	Often abundant
Daily migration of algae	Considerable	Limited
Characteristic algal groups	Chlorophyseae: Desmids *Staurastrum* Diatomaceae: *Tabellaria* Chrysophyceae: *Dinobryon*	Cyanophyseae: *Anabena* *Aphanizomenon* *Microcystis* *Oscillatoria* *rubescens* *Cyclotella* Diatomaceae: *Melosira* *Fragilaria* *Asterionella*
Characteristic zooplankton groups	*Bosmina obtusirostris* *B. coregoni* *Diaptomus gracilus*	*B. longiristris* *Daphnia cucullata*
Characteristic fish	Salmon, trout	Pike, perch
2. *Chemical*		
Dissolved oxygen in hypolimnion	High during whole year	Low or zero during stagnant periods
Water conductivity	Often low	Sometimes very high
3. *Physical*		
Mean depth	Often deep	Often shallow
Volume of hypolimnion	Usually large	Usually small
Temperature of hypolimnion	Usually cold	Usually warm

[1] Adapted from [21-25].

In determining the trophic category of a lake, distinguishing between causes, symptoms and expert judgements is important. In general, the causes are excessive loads of nutrients (usually phosphorus in temperate regions although other factors, such as trace elements, may sometimes be of importance; see section 3.2 of Part 1). Climatological and morphological factors also are of great importance; for instance, the residence time of the water in a lake has a strong influence on the impact of the nutrient load [26]. The symptoms are many and vary from one lake to another (see Table 1). For example, deoxygenation occurs only if the climatological and morphological conditions cause a stratification of the water mass. The expert judgement, finally, encompasses the previous experiences of the expert and also may be influenced by his or her experience-based knowledge of alternative water uses.

In a eutrophication study, the selection of parameters that can be assessed with a reasonable effort is very important. A list of parameters may become overwhelming if all variables of interest are included. A list of essential and desirable parameters are given below [3]. *Essential variables* are those deemed necessary to provide an adequate knowledge of trophic conditions, scope for comparison between water bodies and improvement in the ability to predict change whether as an improvement in water quality, as a result of the introduction of control measures or by deterioration as a consequence of their absence. *Desirable variables* are those appropriate for laboratories with larger resources, or that want to pursue some areas of investigation to a greater extent. This type of measurement supplements the essential data.

1. Physical

   Essential: Temperature, electrical conductivity, light penetration and colour, solar radiation

2. Chemical

   Essential: pH, dissolved O_2, P, N, SiO_2, alkalinity and acidity, Ca^{++} and Mg^{++}, Na^+ and K^+, SO_4^- and Cl^-, total Fe

   Desirable: Other trace elements (e.g. Mg, Mo) and micropollutants (e.g. pesticides), H_2S, CH_4

3. Biological

Essential: Phytoplankton (chlorophyll *a*), oxygen production (primary production, organic carbon)

Desirable: Phytoplankton identification by genera and counting, C^{14} uptake, zooplankton identification and counting.

Relations between the trophic state and specific limiting values of certain water quality parameters have been established on the basis of the results obtained in the International Programme of Cooperation on Eutrophication Control of Continental Waters. These relations may be presented in a "fixed value" form as shown in Table 2. In following the "fixed value" strategy, however, a certain overlap cannot be avoided (i.e. a lake may be classified as belonging to different trophic classes depending on the parameters considered). Another classification strategy can be obtained after a statistical evaluation of the data in Table 2. This classification scheme, called the "open limit" method, is shown in Table 3. Using the latter system, a lake may be regarded as correctly classified if the maximum deviation of any parameter from the geometrical mean does not exceed +/- 2 units.

Table 2. *Proposed boundary values for trophic categories (fixed boundary system*

Trophic category	$\overline{[P]}_{\lambda}$	$\overline{[chl]}$	[max] [chl]	$\overline{[Sec]}^{y}$	$[min]^{y}$ [Sec]
		mg/m^3		m	
Ultra-oligotrophic	$\leq$4	$\leq$1	$\leq$0.5	$\geq$12	$\geq$6
Oligotrophic	$\leq$10	$\leq$2.5	$\leq$8	$\geq$6	$\geq$3
Mesotrophic	10-35	2.5-8	8.25	3-6	3-15
Eutrophic	35-100	8-25	25-75	3-1.5	1.5-0.7
Hypertrophic	$\geq$100	$\geq$25	$\geq$75	$\leq$1.5	$\leq$0.7

Table 3. *Preliminary classification of trophic state in the OECD eutrophication programme*

Variable (Annual x values)		Oligo-trophic	Meso-trophic	Eutrophic	Hyper trophic
Total P (mg/m^3)		8	26.7	84.4	
	$\bar{x} \pm$ 1 SD	4.85-13.3	14.5-49	48-189	
	$\bar{x} \pm$ 2 SD	2.9-22.1	7.9-90.8	16.8-424	
	Range	3-17.7	10.9-95.6	16.2-386	750-1200
	n	21	19 (21)	71 (72)	2
Total N (mg/m^3)	$\bar{x}$	661	753	1875	
	$\bar{x} \pm$ 1 SD	371-1180	485-1170	861-4081	
	$\bar{x} \pm$ 2 SD	208-2103	313-1816	395-8913	
	Range	307-1630	361-1387	393-6100	
	n	11	8	37 (38)	
Chlorophyll *a* (mg/m^3)	$\bar{x}$	1.7	4.7	14.3	
	$\bar{x} \pm$ 1 SD	0.8-3.4	3-7.4	6.7-31	
	$\bar{x} \pm$ 2 SD	0.4-7.1	1.9-11.6	3.1-66	
	Range	0.3-4.5	3-11	2.7-78	100-150
	n	22	16 (17)	70 (72)	2
Chlorophyll *a* peak value (mg/m^3)	$\bar{x}$	4.2	16.1	42.6	
	$\bar{x} \pm$ 1 SD	2.6-7.6	8.9-29	16.9-107	
	$\bar{x} \pm$ 2 SD	1.5-13	4.9-52.5	6.7-270	
	Range	1.3-10.6	4.9-49.5	9.5-275	
	n	16	12	46	
Secchi depth (m)	$\bar{x}$	9.9	4.2	2.45	
	$\bar{x} \pm$ 1 SD	5.9-16.5	2.4-7.4	1.5-4	
	$\bar{x} \pm$ 2 SD	3.6-27.5	1.4-13	0.9-4	
	Range	5.4-28.3	1.5-8.1	0.8-7	0.4-0.5
	n	13	20	70 (72)	

$\bar{x}$ = Geometric mean. SD = Standard deviation.
() = Value in bracket refers to the number of variables (n) employed in the first calculation.

[1] The geometric mean (based on log 10 transformation) was calculated after removing values less than or greater than 2 SD obtained (where applicable) in the first calculation.

3.4 Prediction of trophic state

The first efforts to systematize the classification of water bodies into trophic categories and to establish relations between trophic states and external loads of nutrients were made by Vollenweider during the 1960s and 1970s [1,26,27]. During this work, two important aspects of the problem were recognized: the necessity to establish statistical relations for a large number of lakes and the importance of finding valid relations between the loads, the physical circumstances (such as the residence time of the water in a lake) and the observed trophic status observed for a lake. After these initial efforts, a number of important studies on the subject have been carried out establishing more-detailed trophic relations [3,4].

One objective in the OECD report on eutrophication [3] was to improve the accuracy of the Vollenweider relations between trophic state and nutrient load to establish more-efficient management practices for culturally eutrophied water bodies. Based on data from some 150 lakes and reservoirs in the temperate region, a large number of correlations between indicators of trophic status and nutritional conditions or nutritional loads have been established in the OECD report and are given below:

Correlations between trophic indications and nutritional conditions in lakes; Production-limited factors

Nitrogen versus phosphorus

Orthophosphate-P versus total phosphorus; mineral nitrogen versus total nitrogen

Mineral nitrogen versus orthophosphate-P; preliminary assessment of prevailing nutrient limitation

Correlation between biomass (average peak chlorophyll) versus inlake phosphorus and nitrogen concentrations

Peak chlorophyll versus average chlorophyll

Annual chlorophyll/total phosphorus ratio versus total phosphorus concentration

Transparency versus chlorophyll and versus phosphorus concentrations, respectively

Primary production versus phosphorus and chlorophyll concentrations, respectively

Correlations between trophic indicators and loading

The $[M]_\lambda / [M]_j$ ratio

Correlation between phosphorus and nitrogen loadings and corresponding inlake concentrations

Correlation between trophic parameters and phosphorus and nitrogen loadings, respectively

Primary production versus phosphorus loading

Hypolimnetic oxygen depletion

Regressions between nutritional factors and trophic indicators

Regressions between chlorophyll and inlake phosphorus

Secchi transparency against chlorophyll and inlake phosphorus

Yearly primary production versus phosphorus and chlorophyll, respectively

Regressions between trophic indicators and nutrient loadings.

Both qualitative and quantitative predictions of the development of the trophic state of a lake or reservoir can be made on the basis of the established relations. In particular, the response of a lake to a reduced external nutrient load can be predicted, which is evidently of great importance for the management of eutrophied systems.

In the present context, quantitative predictions signify the forecast of the development of the mean value of a given variable, for example, the chlorophyll concentration, as a function of, say, the phosphorus concentration. Quantitative predictions thus encompass previsions of the changes in the mean values of response parameters, such as chlorophyll or algal production, to changes in the nutrient load.

Qualitative predictions are related to the variability in the response parameters. The mean values evidently do not contain all the information of the system; of equal or sometimes more interest are the maximum values during a season. A qualitative prediction thus contains probability estimates of both mean and maximum values, and the frequency of occurrence of the latter, of a number of response parameters. This information is very important to the manager of the lake or reservoir, and it is thus essential that uncertainties in the predictions are reported and justified (see list above).

4. DIFFERENCES BETWEEN LAKES AND RESERVOIRS

The construction of dams on rivers in order to create artificial lakes is one of the oldest and most common measures used by humans to change the environment. Probably hundreds of thousand or even millions of man-made reservoirs with a surface of more than 1 ha exist today. However, only during the last century have very large reservoirs, with a surface of more than 100 km^2 been constructed. Such large reservoirs give rise to a multitude of effects on the local hydrology and climate, on both the terrestrial and aquatic ecology, and on the social situation on a regional scale. According to a recent estimate, 50 artificial lakes with a surface of more than 1000 km^2 (seven of which are in Africa) and about 300 with a surface between 100 and 1000 km^2 exist in the world today.

When a dam is constructed on a river, the existing terrestrial and river systems are transformed into a lacustrine or stagnant aquatic system. The construction of a large reservoir is thus a gigantic ecological experiment in which the species adapted for a life in a river must now adapt to lacustrine conditions and form a new complex ecosystem with all its connections and interactions. This is not always possible to achieve in a short time period. Some of the original species may, for example, adapt to the new conditions and form stable populations in the reservoir while others may disappear. In general, the species diversity in a young reservoir is consequently smaller than that in a river. Because of the abundance of nutrients coming from the decaying terrestrial vegetation on the bottom of the reservoir, the productivity increases drastically during the first years of the life of a reservoir. The increased productivity stems from easily adapted species - "opportunists" - that multiply and grow in an explosive fashion. At the same time, the new ecosystem shows large fluctuations in biomass and in the relative abundance of different species. During this period, the system is extremely unstable and may "derail", causing massive fish death.

After this initial phase, characterized by a high productivity and instability, which may last for 20-30 years in cold or temperate climates but only for 5-10 years in warm climates, the reservoir usually starts to show a decreased productivity. At the same time, the species diversity increases due to immigration of new species. This evolution may be illustrated as in Figure 2. In many cases a second phase of eutrophication occurs due to an increased nutrient load from villages and farms prospering from the new water resource.

Apart from the low stability of the ecosystem of a young reservoir, a whole series of other phenomena takes place that distinguishes reservoirs from natural lakes. It is here sufficient to mention the physical form of the basin, which is different for reservoirs and (most) lakes. The deepest part of a reservoir is always found at an extremity of the water body, in the immediate vicinity of the dam. This fact, in combination with the usually mountainous surroundings, makes the wind-induced mixing of the water mass different in a reservoir than in a natural lake. If a reservoir is compared with a lake of the same volume and mean depth, in general, the maximum depth in the reservoir is larger than in the lake and, furthermore, the former has a larger stagnant hypolimnion than the latter.

The volume variations in reservoirs differ both in size and frequency from those of lakes. The management of a reservoir evidently gives rise to a large variation of the hydraulic conditions and, consequently, to large variations in the yearly mean depths, water residence times and stratification periods. The bottom gate can be used during well-selected periods for manipulation and, to some extent, for control of the eutrophication processes and the water quality in a reservoir. By discharging large quantities of water during short periods of time, currents can be introduced in the water mass and even provoke a mixing of the hypolimnic and epilimnic waters. This method also can be used to improve the overall water quality by discharging hypolimnic water of bad quality (water rich in hydrogen sulfate and/or nutritional elements).

Because of the important differences between lakes and reservoirs concerning the frequency and vertical distribution of the normal discharges, the application of mathematical models, developed for lakes, to reservoirs must be done with great prudence.

Reservoir eutrophication is discussed in great detail, for example, by Bernhardt et al. [29,30], Nusch [31], Steel [32] and Duncan [33].

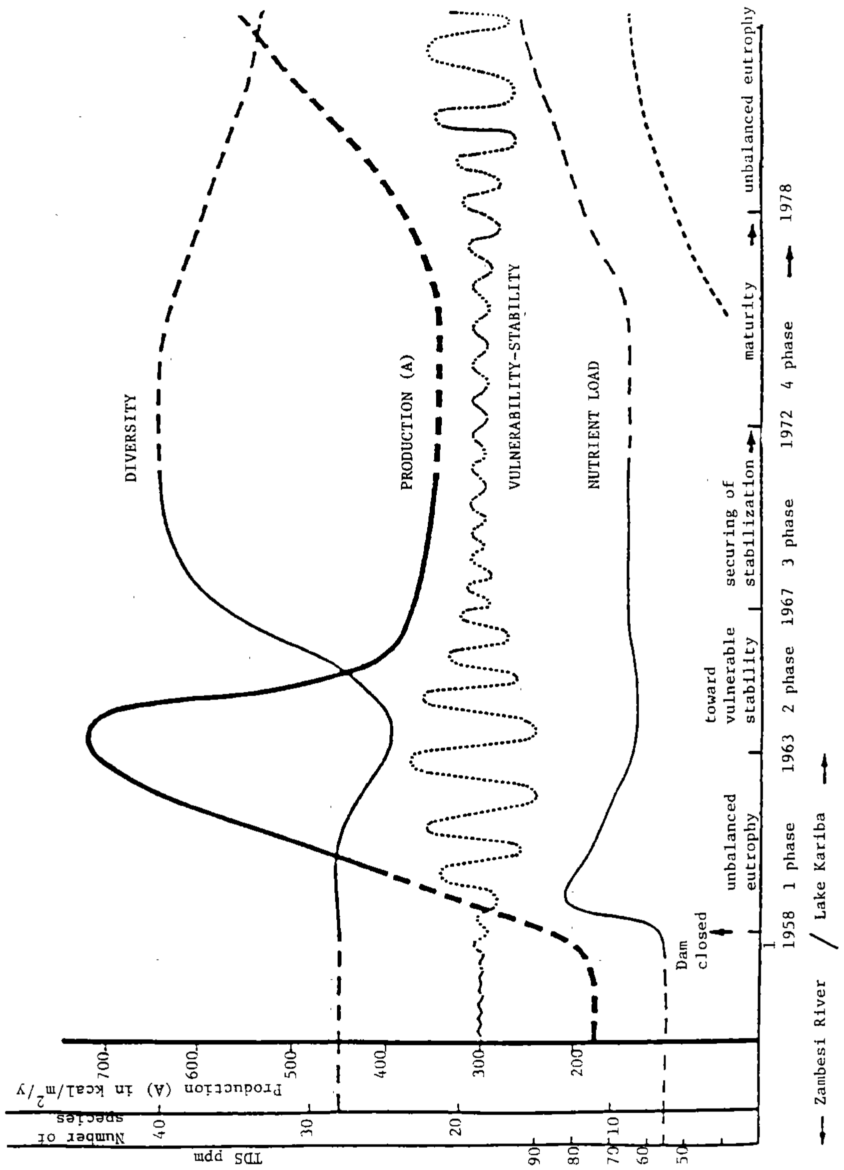


Fig. 2. *The different phases of evolution of Lake Kariba, Zimbabwe, demonstrated in terms of the nutrient load, total production, species diversity and vulnerability-stability of the ecosystem* [28]

5. LAKES AND RESERVOIRS IN WARM CLIMATES

5.1 General characteristics

A strict definition of a warm lake or reservoir is difficult. Several more or less operational definitions have been proposed. At first sight, the concept simply seems to imply lakes and reservoirs in the tropical belt. However, Burgis [34] has pointed out that a purely geographical definition - lakes in the region between the tropic of cancer and the tropic of capricorn - encompasses lakes with very varied characteristics. This zone has warm, shallow and marshy lakes, very large, warm and deep lakes, salt lakes of all sizes and small cold lakes at high altitudes. One must thus be careful with purely geographic definitions. In addition, certain lakes outside the tropical belt can be classified as warm. A more pertinent approach may be to base the definition of a warm lake on the climatic system proposed by Koeppen [35]. Typical warm lakes are, according to Koeppen, found in climates A_f (tropical humid) and A_w (periodically dry, savannas). These lakes are characterized by a mean annual temperature of more than 20 °C with small annual variations. Relatively warm lakes showing large yearly temperature variations are found in temperature zones of type C_s (Mediterranean, dry in the summers) and C_w (monsoon, dry in the winter) and partly also in Bs (prairie) and B_w (desert). Their minimum temperatures are typically larger than 10 °C, but their water temperatures rise to above 20 °C during several summer months. Taking these considerations into account, Vollenweider [8] has suggested a definition according to which warm lakes and reservoirs are those with a minimum temperature above 10 °C, a temperature above 20 °C for more than 6 months per year and (consequently) a mean temperature above 15 °C.

According to this definition, all tropical lakes below an altitude of 2500-3000 m are warm. Lakes and reservoirs in typical Mediterranean climates, which in Europe and the near Orient correspond to regions south of the January 10 °C isopleth or the northern limit for cultivation of olives and citrus fruits, also are warm according to this definition. For the southern hemisphere, warm lakes are found north of the July 10 °C isopleth (see also Table 4).

Another important characteristic, mentioned above, of lakes in the warm belt is the small yearly variations in the water temperature [36]. In general, the water temperature is high, but the mean temperature of tropical lakes at high altitudes may be fairly low. The modest yearly temperature variations are of

Table 4. *Definitions of a warm lake*

Parameter	Sub-Mediterranean	Mediterranean	Subtropical	Tropical (Meso-) trophic	Tropical (Eu-) trophic
Yearly minimum temperature	>5 <10	>10 <15	>15 <20	>20 <25	>25
Oxygen concentration saturation (mg/l)	>11	>10	>9	>8	<8
Relative metabolic activity	1	1.5	2	3	4

course a consequence of the small variability of the insolation and the atmospheric temperature at these low latitudes. In spite of the large insolation intensity in the tropical zone (140-220 kcal/cm^2/y in comparison with 80-100 kcal/cm^2/y at the 40° latitude), the fairly constant radiation over the year makes the limnological processes relatively inefficient. In the tropical zone, the precipitation and the seasonal winds play a decisive role in this respect [37]. The small seasonal variation in the atmospheric temperatures in the tropical zone explains the limited variation in the seasonal water temperatures. It also explains the small values of the annual heat balance typical for tropical lakes in comparison with temperate lakes (Table 5).

5.2 Stratification and mixing of the water mass

The thermal conditions in the water mass form the principal factor determining the characteristics of the water mixing in lakes and reservoirs. The extent and frequency of the mixing are decisive for the regulation of the production dynamics, the transport of nutrients, the oxygen concentrations, etc. The different characteristics of the mixing of the water mass in lakes and reservoirs are shown in Table 6.

Knowing the thermal conditions and mixing characteristics of warm lakes is even more important than for temperate lakes. The

Table 5. *Annual heat balance in some lakes*

Lake	Latitude	Altitude (m)	Heat balance (cal/cm^2)	Reference
Victoria (E. Africa)	1°00'S	1134	10 000	[38]
Klindungan	7°40'S	10	3 400	[39]
Lanoa (Philippines)	8°00'N	720	4500-7250	[40]
Guija	14°13'N	426	5 410	[41]
Amatitlan	14°25'N	1189	8 510	[41]
Atitlan	14°40'N	1555	22 100	[41]
Titicaca (Peru/Bolivia)	16°00'S	4000	18 900	[42]
Kinneret (Israel)	32°42'N	-209	44 000	[43]
Tahoe (USA)	39°00'N	–	34 800	[41]
Léman (Switzerland/ France)	42°34'N	263	32 300	[41]
Michigan (USA/Canada)	44°00'N	176	52 400	[41]
Baikal (USSR)	53°00'N	453	65 500	[41]

reason is that the vertical transports and mass exchanges generated by the mixing process are in general more subtle and more directly connected with the biological response in warm lakes than in temperate lakes. Consequently, the ability to forecast the thermal conditions in a projected reservoir in the warm belt is of primordial importance if the future biological processes in the reservoir are to be understood and the risks for a future eutrophication evaluated.

In lakes that are warm according to the above definitions, the mixing characteristics are as varied as their thermal conditions. One can thus encounter cases ranging from holomictic to meromictic conditions, including oligomictic, monomictic and polymictic lakes. The characteristic mixing in a particular case depends on the geographical situation, the local

Table 6. *Mixing characteristics of lakes and reservoirs*

Type of extension	Type of mixing	
Spatial	Holomixis	- Mixing of the whole water body
	Meromixis	- Mixing of the water mass close to the surface; deeper water layers always remain more or less stagnant
Temporal	Amixis	- No mixing
	Oligomixis	- Mixing occurs in an irregular fashion and varies for different years
	Monomixis	- Mixing occurs seasonally once per year
	Dimixis	- Mixing occurs seasonally twice per year
	Polymixis	- Mixing occurs several times per year at irregular intervals

climate, the evaporation, the exposure to winds, and the size and the morphology of the lake. Saline water entering the water mass also may complicate the mixing process.

In the classic studies by Ruttner [44,45], the polymictic characteristics were recognized as fundamental to the particular case depends on the geographical situation, the local understanding of tropical lakes. Recent studies have shown a rich variety of mixing situations in warm lakes. The frequencies of lakes being mixed daily, seasonally or irregularly seem to depend in particular on the size of the lake, the mean depth, the influence of the wind and the relative humidity of the air [46]. The effect of the mean depth on the mixing is illustrated by the differences between Lake George (Uganda) and Lake Victoria (E. Africa), both of which are exposed to equatorial conditions. Lake George (mean depth 2.4 m) shows daily stratification-mixing cycles. The thermal stratification created during the day is rapidly destroyed during the night. No stable thermocline thus develops in this lake [34]. On the other hand, Lake Victoria (mean depth 40 m) shows a relatively stable stratification (with a superimposed polymixis in the epilimnion) between January and June, followed by the disappearance of the thermocline in July/August [47]. Baxter et al. [46] suggest that at least three stratification types occur in small- and medium-sized African lakes at medium altitude:

a. In lakes with a maximum depth of 15-30 m, complete mixing is normal and stratification occurs twice a day (polymictic conditions) - for example, Lakes Aranguadi, Tana, Zwei, Naivasha, Mugesera, Bilila, Sake and Rugwera.

b. In small lakes (with a surface area less than 60 km^2) with steep slopes protected from the wind, infrequent mixing (oligomixis) or absence of mixing (meromixis) is normal - for example, Lakes Bunyoni, Ndalaga, Luhondo and Paivlo.

c. In deep lakes with gradual thermal gradients, a complete mixing may occur occasionally - for example, Lakes Shala, Langano, Awassa and Bulera.

In deep lakes, the stratification normally persists for a part of the year. Very deep tropical lakes, such as Lake Tanganyika in E. Africa and Lake Nyasi (formerly Malawi) in Malawi (with medium depths of 700 and 426 m, respectively), show a permanent stratification (amixis and/or meromixis). The mean depth and the dimension of the hypolimnion play a major role in the persistence of the meromixis of these lakes. In other cases, such as Lake Kiwu, the inflow of saline water may be responsible for the stratification [8].

Vertical temperature profiles in warm lakes often show very small temperature differences between the surface and the bottom. For Lake Victoria and Lake Albert (Uganda/Zaire), the temperature difference between the epilimnion and hypolimnion is only about 1 °C. At high temperatures, the density difference as a function of temperature is much larger than at low temperatures. Consequently, one finds the most stable stratifications between the latitudes at 20° and 30° [36]. As an example, one may cite Lake Nasser (22°N) (Egypt) and Lake Kinneret (32°N) where the density differences between the epilimnion and hypolimnion is 4.67 and 3.75 g/l, respectively. Closer to the equator, this stability decreases because of the reduction of the thermal gradient during the year (Lake Albert has a maximum density difference of only 0.34 g/l). When one approaches the poles, the stability decreases because of the lower temperatures of the water (for Lake Léman the density difference is only 1.87 g/l in spite of a temperature difference of 16°C between the surface and bottom waters).

The speed with which the surface water, which is cold during the winter, is heated during the spring is of paramount importance for the development of a vertical stratification and for the isolation of the epilimnion. A strong and rapid heating of the surface gives rise to the formation of a shallow thermocline that prevents the deep layers from warming up. The consequence of this is that one often finds thermoclines close to the surface and strong temperature gradients at high latitudes. To the contrary, at low latitudes where the annual temperature variations are limited and where consequently the heating rate of the surface is low, one finds weak thermoclines and thus lakes that are easily mixed by the wind [36].

Most lakes in the tropical belt develop a deeper epilimnion than lakes in temperate regions. In Lake Lanao (8°N with a maximum depth of 112 m), the thermocline is 40 m below the surface, and in Lake Tanganyika, the thermocline is found at a depth of 60 m. In Lake Léman (45°N, maximum depth 310 m), the thermocline is situated at a depth of 15 m in spite of prevailing strong winds. Two types of situation may occur when a deep epilimnion has developed in a warm lake: the daily winds may cause a mixing of the whole epilimnion or the epilimnion proper may remain stratified during significant time periods. In the latter situation, wind-induced mixing of the deeper parts of the epilimnion will occur only occasionally. This partial mixing of the epilimnion causes the appearance of several thermoclines, a typical situation for many lakes in the tropical region, particularly in areas where the winds are irregular. The daily temperature variations are often more pronounced than the seasonal temperature variations in warm climates. This situation causes a microstratification in the shallowest water layers. In shallow lakes this microstratification, which appears in the afternoon, is often destroyed during the night. The effect of these daily variations is an efficient nutrient transport between the bottom and the surface in microstratified lakes, which explains the occurrence of permanent water flowering in lakes such as Lake George and Lake Tchad. In deep lakes, the primary production normally occurs in the surface layers of the epilimnion (the euphotic zone). The deeper layers of the epilimnion (layers still situated above the thermocline) are characterized by an accumulation of suspended organic matter and nutrients and, as a consequence, of deficits of dissolved oxygen. With certain intervals, when the whole epilimnion becomes mixed due to occasional strong winds, the difference in chemical composition in different epilimnic layers causes an acceleration in the primary production in the euphotic zone. Lewis [48] has suggested the term "atelomixis" for describing this occasional mixing of chemically different water masses.

This also includes water movements across the thermocline during the warm season, which causes a transport of nutrient-rich hypolimnic water into the epilimnion. Atelomixis is a phenomenon that is much more frequent in tropical regions, where the lakes often have a deep epilimnion, than in the temperate zone. In warm lakes belonging to the Mediterranean or subtropical category (i.e. in climatological zones with an important variation in seasonal temperatures), a quite particular situation may occur. An example of this is provided by the Sidi Mohammed Ben Abdellah reservoir in Morocco. In this case a very strong thermocline is formed for the following reasons. The first is the climatological situation in the region, with fairly cold winters and hot summers with an intense insolation. The second is the fact that almost all the inflow to the reservoir occurs during the cold season. The third is the relatively weak and variable winds during the summer. The strong thermocline is formed at a depth of 12 to 15 m, and the temperature difference between the epilimnion and the hypolimnion is about 10 °C during the summer months (15–16 °C in the hypolimnion and 25–28 °C in the epilimnion).

This situation gives rise to a stagnant layer of water close to the thermocline, the metalimnion. This layer contributes to the very efficient isolation of the hypolimnion from the epilimnion by which practically no water is exchanged between the two major water bodies during the stagnant season. The strong density gradient in the metalimnic layer forms a serious obstacle against the penetration of detritus particles sedimenting out of the euphotic zone.

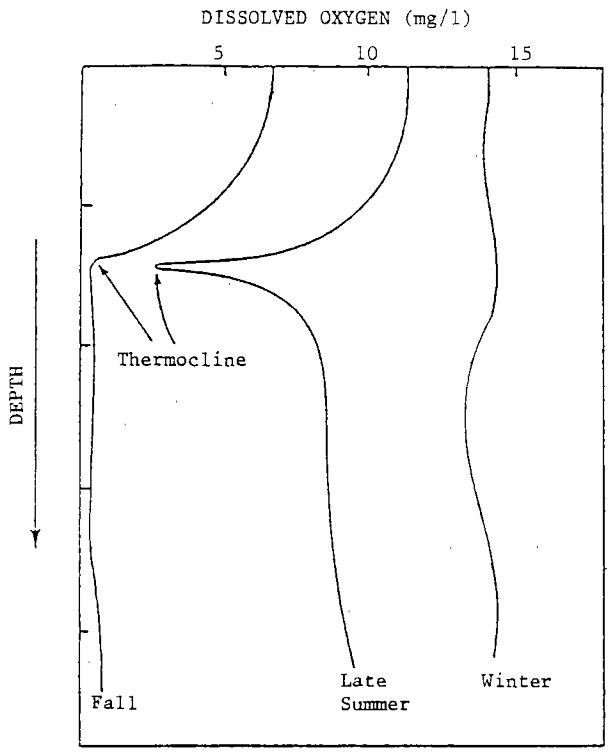
Another example of this kind of situation is provided by Lake Kinneret, where only 5–10% of the carbon fixed by the photosynthetic process reaches the sediments [49]. Very stable thermoclines also may form in temperate lakes, but there are two principal differences between those situations and corresponding situations in warm lakes:

1. Warm lakes generally have a deeper epilimnion, implying a longer residence time for the detritic particles in the euphotic zone.

2. Higher water temperatures in the warm lakes imply higher metabolic rates.

One direct consequence of a very accentuated thermocline is the establishment of a bimodal dissolved oxygen profile caused by the decomposition of organic material accumulated in the metalimnion (see Fig. 3).

Fig. 3. *Curves showing the dissolved oxygen concentrations as a function of depth and season*



Typical characteristics of Mediterranean and subtropical lakes in comparison with temperate lakes are thus (in addition to the phenomenon of atelomixis in tropical lakes):

- increased rate of liberation and recycling of nutrients essential for the algal production in a zone in contact with the euphotic zone;

- increased residence times for the nutrients in the epilimnion, that is, a decreased loss towards the hypolimnion (in this context it should be recalled that orthophosphate liberated from organic compounds does not form insoluble complexes with Fe, Al or Mn under anoxic conditions and that, as a consequence, is not easily precipitated out of the metalimnic zone [50]);

- maintenance of a high production in the euphotic zone;

- decreased consumption of oxygen in the hypolimnion.

In conclusion, warm, deep lakes and reservoirs show certain aspects that make them resemble shallow nonstratified temperate lakes: namely, the high degree of recycling of nutritional elements and the ensuing prolongation of the production period. This contrasts to the situation in temperate regions where the differences between deep and shallow lakes are very clear in this respect. A deep temperate lake usually suffers from an extreme deficit of nutrients in the euphotic zone stemming from the establishment of a stable thermocline in the spring.

5.3 Interactions with the drainage basin - dissolved and particulate loadings

Section 2 pointed out that all the processes in a lake, including the development of the lake as a whole, depend directly on the characteristics of the drainage basin. As a direct consequence, geolithic factors, vegetation cover, slopes, etc. in the drainage basin play a very important role in determining the conditions in the lake. It is thus important to find out if any general characteristics of drainage basins in warm climates differ from those in temperate or cold climates. One difference which is reasonably general is undoubtedly that the soils in the tropical and subtropical zones are more exposed to erosion than soils in temperate or cold areas. This fact is a consequence of the climate, and entails among other things less organic material in the soil and a thin humus cover. The heavy rains which are frequent in the tropical and subtropical zones thus often give rise to large quantities of suspended material in the rivers, in particular during the flood season.

As an example, let us compare a drainage basin in northern Europe with a drainage basin in north Africa, both subject to the same yearly precipitation. The vegetation cover in northern Europe is much more important than that in north Africa, and the distribution of the precipitation over the year is very different in the two regions. The precipitation in northern Europe is fairly evenly distributed over the year and, moreover, during the winter it arrives in the form of snow. In North Africa, on the other hand, the yearly precipitation usually arrives within a few weeks. This latter situation entails that the soil is extremely desiccated when the rainy period starts and that the run-off becomes violent, washing off large amounts of inorganic particles of different sizes as well as material in suspension originating from the soil. What are the consequences of these differences between the climatic zones for the productivity - and for the sensitivity to eutrophication - of the lakes and reservoirs in the two zones? In other words, are the models describing the relations between nutrient load and productivity or biomass, models that have been developed for lakes in temperate or cold climates, directly applicable to lakes and reservoirs in warm climates? It is, as always, difficult to generalize. Every combination of lake and drainage basin has its own individual characteristics. We will illustrate this point with some examples from the subtropical semi-arid zone:

1. Lake Burley Griffin in Australia: surface, 700 ha; volume, 33×10^6 m^3; mean depth, 4.7 m; total phosphorus load, 9 $g/m^2/y$; phosphorus concentration 52 mg/m^3; chlorophyll *a* concentration, 3.0-3.3 mg/m^3 [51]. According to the Vollenweider model [27], the chlorophyll *a* concentration should have been about 23 mg/m^3. Rosich & Collen [52] believe that this difference is due largely to the grazing by zooplankton.

2. The Mount Bold reservoir in Australia: surface, 225 ha; volume, 30×10^6 m^3; mean depth, 13 m; total phoshorus load, 6 $g/m^2/y$; phosphorus concentration, 100-180 mg/m^3; chlorophyll *a* concentration,3.0-3.3 mg/m^3 [51]. The predictions by the Vollenweider model and by the model suggested by Lee et al. [13] give values for the chlorophyll *a* concentration of 24 and 19.6 mg/m^3, respectively. The influence of the turbidity of the water and competition of the nutrient resources have been suggested as explanations for this difference.

3. Lake Kinneret in Israel: surface, 16 800 ha; volume, 4300×10^6 m^3; mean depth, 25 m; total phosphorus load, 0.42-1.7 $g/m^2/y$. Serruya [53] has pointed out the linear relation between the nutrient load and the algal biomass during the bloom the next year for biomasses smaller than 140 g of algae per square meter in this lake. For larger biomasses, factors other than the nutrient load, such as light penetration, excretion pro-ducts and (in particular at very high loads) a portion of the phosphorus remaining in a bound, inaccessible form start to become important for limiting algal production.

4. The Sidi Mohammed Ben Abdellah reservoir situated on the Bou Regreg River in Morocco: surface, 3000 ha; volume, 500 x 10^6 m^3; mean depth, 17 m; total phosphorus load, 170 $g/m^2/y$ (1979) and 9 $g/m^2/y$ (1980). In spite of the difference in total phosphorus load between the 1979 and 1980 [54, 55], the primary production and the algal biomass were practically the same [56,57]. The very large values for the phosphorus load in 1979 coincided with an extremely large flood which gave rise to a high turbidity followed by a fast sedimentation of phosphorus in the reservoir. This sedimentation was followed by a flocculation of the phosphate in solution, presumably in the form of hydroxyapatite. The net effect of these two processes was that in spite of the high phosphorus load only 15 mg/m^3 of orthophosphate was found in the surface water in the spring of 1979. In this case an extreme load of phosphorus did not give rise to a corresponding response in productivity because of the appearance of "autoregulating" factors.

5. The Sidi Salem reservoir on the Medjerdah River in Tunisia: surface, 3000-4000 ha; volume, 350-550 x 10^6 m^3; mean depth, 12-13 m; total phosphorus load, about 1 $g/m^2/y$. According to the Vollenweider model, this phosphorus load is excessive for Sidi Salem and should result in eutrification. Nevertheless, the mean concentration of chlorophyll *a* is presently only about 4-6 mg/m^3 and the mean Secchi disc depth is about 1.5 m with maxima of 4.3 m [30,58]. Thus, Sidi Salem does not show any typical eutrophication symptoms, which may be explained by the very high concentrations of calcium (100-200 mg/l) facilitating a transformation of the phosphates into nonsoluble phosphorus and thus a precipitation of phosphorus from the trophogenic zone.

These few examples show the importance of the characteristics of the drainage basins, in particular the composition of soils that are eroded and transported to the watercourses. Since the transport of both particulate and suspended matter to lakes and reservoirs often is much more important in warm than in temperate or cold climates, the external factors connected with the drainage basin should be studied in great detail in the tropical and subtropical zones.

5.4 Biomass, primary production and recycling of nutrients

The following discussion of the productivity of lakes and reservoirs in warm climates and of various regulating factors under such conditions concentrates on deep lakes and reservoirs, where a stable stratification occurs at least during part of the year. This accentuation of deep lakes and reservoirs has been chosen partly because the deep water bodies in particular show differences between the temperate and the warm zones, thus meriting indepth studies, and partly because artificial lakes, usually reservoirs, usually are very important both socially and economically.

In this report, the discussion of biomass and primary production in lakes is limited to phytoplankton. Even in this case, however, data from the warm zones are scarce, in particular for the biomass. A more commonly measured parameter is chlorophyll *a*, but such data do not always give a fair picture of the algal biomass since the chlorophyll content varies substantially between taxonomic groups. However, a few lakes in the warm zone have been fairly well investigated; some examples are shown in Table 7 (after [36]). The biomass, measured in g/m^2, varies quite drastically among the lakes studied: from 7.4 g/m^2 in Lake Valencia in Venezuela to 24 g/m^2 in Lake Lanao in the Philippines to 40–80 g/m^2 in Lake Kinneret in Israel.

In spite of the sometimes low biomass, the photosynthetic activity is usually very high in warm lakes compared to temperate and cold lakes. This point is illustrated in Table 8

Table 7. *Algal biomass in some warm lakes*

Lake	Biomass (mg/m^3)	Chlorophyll (mg/m^3)
Victoria	–	1.2–5.5
Titicaca	130–280	–
Tanganyika	60–160	0.1–4.5
Lanao	5000–6000	–
Kinneret	500–18 000	10–148

Table 8. *Net annual production in selected lakes*

Lake	Insolation (10^6kcal/m^2/y)	Net production (gC/m^2/y)	Reference
Victoria	1.54	640	[47]
Kinneret	1.3	635	[59]
Lanao	1.49	620	[48]
Sylvan	1.2	570	[60]
Titicaca	1.45	530	[61]
Bou Regreg	1.3	400	[15]
Minnetoka	1.1	300	[62]
Tanganyika	1.5	290	[63]
Clear	1.2	160	]64]
Michigan	–	145	[65]
Baikal	–	123	[66]
Erken	0.83	104	[67]

where the net annual production for a number of tropical, subtropical and temperate lakes is shown.

With the exception of Lake Tanganyika, the great tropical lakes have a much higher production than the temperate lakes. Lake Minnetonka in Minnesota and Lake Sylvan in Indiana are both considered hypertrophic and Clear Lake in California and Lake Erken eutrophic. Both Lake Victoria and Lake Lanao have a much higher productivity than any of the temperate lakes, yet none of the warm lakes considered displays typical symptoms of eutrophication, such as reduced transparency, high biomass or high concentrations of nutrients in the water. The annual production of Lake Victoria and Lake Lanao is in fact about 5–10 times higher than the annual production in temperate lakes having equivalent concentrations of nutrients and similar transparencies. The insolation for the two tropical lakes is about twice the insolation at the 45th latitude [4].

Let us return to the exception among the tropical lakes in Table 8: Lake Tanganyika. In spite of the fairly low value of the net production measured as fixed carbon per unit surface per year, this lake has the highest multiplication rate of phytoplankton recorded for any tropical lake [63]. The multiplication rate has been estimated at 0.9/d by Peterson [68], and this value is about twice that estimated for Lakes Titicaca and

Lanao. A multiplication rate as high as that observed in Lake Tanganyika, in combination with the relatively low values for the biomass in the lake (the annual mean value of the biomass is 140 mg/m^3 in wet weight and 1.2 mg/m^3 chlorophyll in the euphotic zone according to Hecky & Kling [69]), indicate that the loss rate also must be very high. The high multiplication rate and the high loss rate signify that the ecosystem in this very old lake must have a very high trophic efficiency (i.e. the energy fixed by the algae must be efficiently transported up through the trophic levels and become accumulated in fish biomass rather than lost as detritus at the bottom of the lake).

This type of behaviour seems to be characteristic for many ecosystems in the tropical zone. Thus, the growth rate is almost always much higher in warm lakes than in lakes in the temperate zone. How can the high annual production in the tropical lakes be explained if it is not caused by the high insolation? Let us return to the question of the high degree of nutrient recycling. Scavia [70] has analysed the rate of recycling of phosphorus in the epilimnion in Lake Ontario (on the USA-Canada border) during the stratification period. His results are shown in Table 9.

The estimated rate of recycling of phosphorus through mineralization of detritus represents a regeneration of about 2%

Table 9. *Recycling rates of phosphorus in the epilimnion of Lake Ontàrio* [70]

Process	% of total recycling
Excretion of phytoplankton	32.8
Excretion of zooplankton	
Herbivores	27.3
Omnivores	10.2
Total	37.5
Decomposition of detritus	17.0
External load	3.2
From the hypolimnion	9.5

of the detritic phosphorus per day. The importance of the zooplankton in the recycling process entails that at the same time as the zooplankton tend to reduce the biomass through grazing pressure, they contribute considerably to the algal production through their recycling of phosphorus. The recycling rate of the nutrients depends on the metabolic rates of bacteria, protozoa and zooplankton. The recycling consists of three distinct processes [4]:

a. the degradation rate and the consumption of organic matter in the euphotic zone;

b. the degradation rate and the consumption of organic matter in the epilimnion under the euphotic zone;
c. the frequency and efficiency of exchanges of matter between the water in the euphotic zone and the deeper water layers.

The degradation rate in the euphotic zone can be expected to be more efficient in warm than in temperate waters simply because metabolic rates generally increase with the temperature. The efficiency of points b and c is connected with the process of atelomixis, which occurs mostly in stratified lakes in the warm zone. There are thus several reasons to believe that the recycling of nutrients is much more efficient in warm lakes than in lakes in the temperate zone.

5.5 Productivity and temperature

Many proposals have been made in the literature that the high productivity in warm lakes is a direct result of the increased temperature on the photosynthetic process. Lewis [48] has, however, pointed out that this interpretation has at least two weak points. First, conclusions drawn from observations of a particular algal community about the temperature dependence of the photosynthesis cannot be directly extrapolated to algal communities adapted to higher temperatures. Second, nutrient and light limitation cannot be corrected by higher temperatures.

In fact, the photosynthetic rate per cell does not seem to change very much with the temperature. It has been reported several times that a change in the temperature produces an initial change in the photosynthetic rate but after temperature adaption, the rate returns to levels close to the original one [71,72]. The adaption of the algae to lower temperatures is reflected by an augmentation of the protein ARN in the cells

[73]. An increased need for nutrients at low temperatures also has been observed.

Although adaption does not produce identical effects at all temperatures, it does tend to reduce the role of temperature as a regulating factor for the metabolic rate of algae. Limitations imposed by the insolation and the nutrients also tend to diminish the effect of the temperature on the regulating mechanisms for the annual primary production.

5.6 Production and trophic efficiency

The symptoms of eutrophication, such as water flowering, excessive growth of macrophytes, high turbidity, bad taste and odours, and deoxygenation of the hypolimnion, may perturb the equilibrium in a lake and, in particular, a reservoir, which often has a young ecosystem which is far from having reached its maturity even under normal circumstances. This phenomenon can occur both in warm and temperate reservoirs, but the consequences are often more serious in warm areas. The following examples from the subtropical zone illustrate this point.

The Sidi Mohammed Ben Abdellah reservoir on the Bou Regreg River has many characteristics in common with Lake Kinneret. The two water systems belong to the same climatic zone, they are both monomictic, they have almost the same mean depths and they have very similar flood characteristics, with most of the nutrients arriving during a short winter period [53,74,75]. A very large fraction of the total phosphorus load (>90%) is removed from the water mass through sedimentation in both lakes, and most of the nitrate (>50%) is used for oxidation reactions (i.e. denitrification). A very striking similarity between the two lakes is that the algal biomass consists of *Peridinium cinctum* (80-90% during January-June in Lake Kinneret and during April-August in Sidi Mohammed Ben Abdellah). The biomass reaches levels of 40-50 g/m^3 in Lake Kinneret [53] and 25-30 g/m^3 in the reservoir on the Bou Regreg River [57]. The Peridiniae are very rich in carbohydrates; the large cells (20 000-120 000 um^3) have a carbon content corresponding to 20% of the wet weight and a chlorophyll content of 0.3% [76]. The elementary composition, C:N:P, varies between 250:18:1 and 650:2:1, which shows that the demand of nitrogen and phosphorus is much smaller for *Peridinium* than for *Chlorophyses* and *Cyanophyses* per unit of assimilated carbon. *Peridinium* has the property of being able to migrate vertically in the water. These algal species rest on sediment surface in the form of cysts until periods of homothermy when they are resuspended into the water and float up to the surface where they can initiate an algal bloom if the conditions are right. During the production period, the cells migrate daily in and out of the euphotic zone and can in this way optimize the use of light and nutrients. The problem associated with this dinoflagellate from the eutrophication point of view is that the grazing pressure exerted by zooplankton is very low. The only consumers of *Peridinium* are the fish species *Tilapia galilea* in Lake Kinneret [77] and *Mugil saliens* in the reservoir on the Bou Regreg River [78]. Since the grazing pressure on the *Peridinium* is low, the trophic chain is directed towards detritus and a large reservoir of organic material is formed in both lakes. However, although only some 10-20% of the organic matter produced during the bloom in the spring is consumed by the herbivores, a large fraction of the organic matter is decomposed before it reaches the hypolimnion. This process is catalysed by microflagellates (Protozoa), which stimulate the hydrolytic bacteria [79]. It should again be underligned that an efficient recycling of nutrients in the epilimnion, which thus is the case in Lake Kinneret, prolongs the period of high primary production and accentuates the symptoms of eutrophication.

Another example of this mechanism is provided by the hypertrophic, monomictic and warm Hartebeespoort reservoir constructed in 1923. Hartebeespoort has a surface of 20 km^2, a volume of 200 Mm^3 and a mean depth of about 10 m. During the early days of this reservoir, the phytoplankton community was dominated by *Peridinium cinctum*, but after 30 years the reservoir developed the characteristics of an "oxidation lagoon". During the last 10 years, water blooms of a toxic species (*Microcystis aeruginosa*) have occurred at regular intervals, as well as a strong increase of the water hyacinth *Eichhornia crassipes* [80].

The chlorophyll concentrations in Hartebeespoort reservoir exceed 240 mg/m^2 for more than 50% of the year and have attained maximum values of 240 mg/m^3. The water blooms have been accompanied by the formation of a thick foam of *Microcystis*. The relation between the biomasses of zooplankton and phytoplankton is in general less than 0.1, which indicates a very inefficient grazing. *Microcystis* are well known to be a very poor nutrient source for zooplankton. The occurrence of *Microcystis* blooms and the strong growth of *Eichhornia*, which is almost useless as a source of energy for higher trophic levels, show quite conclusively that the trophic chain in the Hartebeespoort reservoir is based on detritus. The situation thus accentuates eutrophication symptoms.

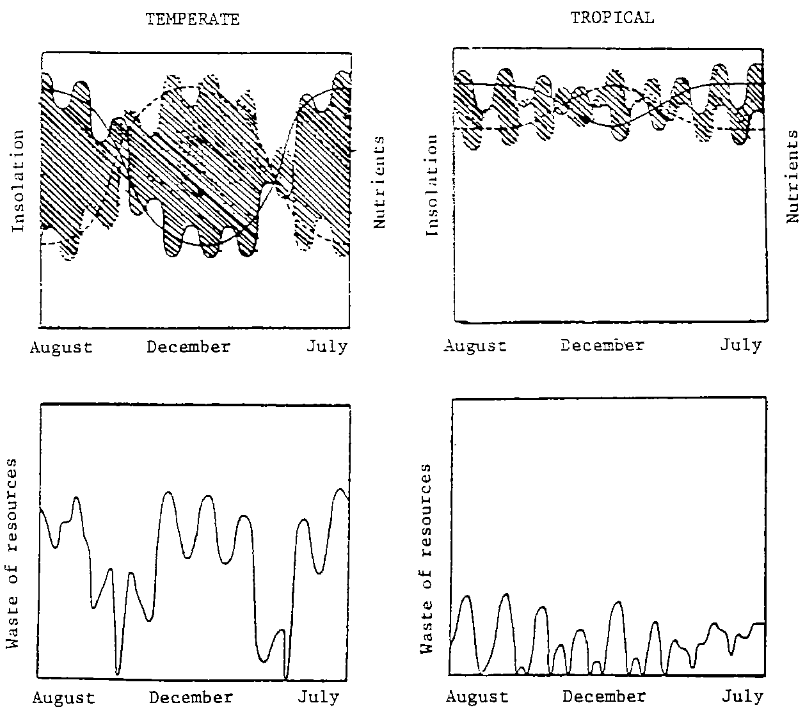
5.7 Resource utilization

In general, the key factor distinguishing lakes in the warm zone from lakes in the temperate zone is the variability in the resources. The productivity in a temperate lake and in a warm lake are, in principle, limited by the availability of nutrients and light. The contrast between the two types of lake does thus not originate from different control factors but rather from the mode of operation of the same type of control factors. The total provision of yearly limiting resources can thus be considered as a measure of the maximum potential productivity. The maximum productivity can be attained only if the resources are distributed in time such as to minimize the differences in mean accessibility at any instant during the year. In this sense the waste of the resources may be defined as the difference between the actual limiting resource at a given instant and other potentially limiting resources integrated over the whole year. The temporal distribution of the limiting resources is schematically illustrated in Figure 4 for a hypothetical situation in a warm and a temperate lake (after Lewis [48]). The waste of resources is larger in the temperate lake both in relation to the total resources and in absolute terms. In the tropical lake the nutrients are more efficiently used because of the more even distribution of the insolation, which permits a more efficient recycling of the nutrients.

5.8 Conclusions

The symptoms of eutrophication, such as water blooms, reduced water transparency, formation of algal foam, bad taste and odours, deoxygenation of the deep layers and/or strong increase in macrophyte biomass, may be regarded as a loss of energy originally fixed by the photosynthesis (i.e. a waste of produced resources). This inefficiency of the ecosystem is directly connected with a fundamental disequilibrium of the system. Put simply, it is the question of a bad harmonization between the different processes in the ecological system. The discussion in the previous sections leads to the conclusion that no fundamental differences probably exist between the processes of eutrophication in warm and temperate lakes. The principal factors causing eutrophication are no doubt the same in the two cases, but their mode of operation is different. In this sense no definitive limits are found between the climatic zones but rather different tendencies between different regions. The use of the resources in tropical lakes is certainly more efficient than the use of resources in temperate lakes. The ecosystems in

Fig. 4. *Schematic representation of temporal variations of the limiting resources in a temperate and a tropical lake, and of the resulting waste of resources*



the warm zone are more equilibrated than in the temperate zone, largely because the former systems in general are older and thus have had time to reach a higher level of maturity. In contrast to this "natural" situation, the evolution of young ecosystems, such as the ones encountered in reservoirs, lack an ecological maturity and are in general extremely unpredictable, in particular in the warm zone. The evolution is unpredictable because the exceptions from the established cause-effect relations are numerous in the warm zone and they play an important role in disequilibrating the systems. The direct implication of this argumentation is that, although the fundamental relations proposed by Vollenweider between nutrient loads and nutrient concentrations on one hand and the productivity in a lake on the other in principle remain valid, these rules have to be applied with a great deal of caution to artificial lakes in the warm zone.

The high degree of unpredictability characterizing the evolution of reservoirs in the warm belt merits deeper studies on individual cases to understand better the mechanisms and the factors giving rise to exceptions to the basic Vollenweider-type rules.

6. USE OF MATHEMATICAL MODELS

6.1 Introduction

Many mathematical models have been developed and applied to a variety of problems in eutrophication studies. In the present report the scope has been narrowed to the lakes; the drainage basin, important as it may be, is dealt with only fairly superficially. One category of models not further covered in this report is that of drainage basin models, which in certain situations may be very useful in providing information on nutrient loads and different nutrient sources. Instead, the focus will be on models of the water body, a category of models that has proven to be indispensable in the evaluation of eutrophication phenomena and for planification and management of the aquatic resources.

Lake and reservoir models can be used for two different but closely related motives. On one hand, they may facilitate the comprehension of the processes, and the interplay between the processes ultimately determining the entire behaviour of a lake. On the other hand, they may serve as a tool for predicting the response of a lake to a change in the resources, such as a modification of the nutrient load. Even in a small lake, it is impractical to study all processes of importance simultaneously. Neither can one make field studies of the effects of changing the values of selected variables (e.g. the phosphorus load). Consequently, a method is strongly needed that allows a simultaneous evaluation of all the accessible information in order to comprehend the processes in the entire lake [81]. The best method to date to reach this goal is to introduce mathematical representations of the lacustrine ecosystem and the interactions between its different parts. This method produces a tool for synthesizing all the known mechanisms and interactions and for simplifying the system. Results obtained from a mathematical model can never replace information from experimental field studies and observations. Instead, since the model is capable of treating simultaneously the information from several different interacting components, the model furnishes a more comprehensive picture than what can be obtained from individual observations. Thus the relative importance of different processes controlling a certain component of the system can be extracted, as well as temporal variations. A model is in fact a hypothesis-testing instrument, and it can be used for investigations of areas where supplementary observations would otherwise be necessary; a model does not create new information but facilitates a quantitative analysis of the whole system [81]. Riley [82] offers three principal approaches for developing mathematical models:

- to deduce equations from statistical correlations of observed data (empirical models);

- to develop mathematical expressions from theoretical considerations;

- to combine these equations with parameters derived from observations (conceptual models).

In the empirical models, the system variables are connected with input variables without a detailed knowledge of the mechanisms in the system. The relations between the trophic state, the mean depth and the phosphorus load for a lake developed by Vollenweider [1] is one example of an empirical model. The conceptual models, on the other hand, describe the interactions between the different components of the system; examples of this kind of model are provided by the mass balance models for nutrients which also have been developed by Vollenweider [26,27,83]. These models use a differential equation to describe the mean concentration of a nutrient as a function of a number of input parameters to the lake. In practice, however, the empirical and the conceptual approaches are often used simultaneously in the model development. We will therefore use a different classification in the present report, namely, simple models, simulation models (dynamic) and ecological models (complex). A more through discussion of different aspects of lake models is given in Rosich's work [81].

6.2 Simple models

The main characteristic of models belonging to this category is that they are indeed very simple. Using the terminology given above, they may be either empirical or conceptual. A linear relation between the total phosphorus concentration during the spring in a lake and the mean chlorophyll concentration during the summer was described for the first time by Sakamota in 1966 [84]. Given that phosphorus is the limiting factor for production (i.e. N/P > 12), one obtains the following relation:

log chl = 1.583 x (lop TP + 1.134) (r = 0.98)

There are several modified models, for example, the expression proposed by Verduin et al. [85]. In the Verduin equation, the mean depth of the lake (Z) and the depth of the euphotic zone (E) are accounted for, the reason being the inadequacy of the original formulation to accommodate lakes with a high inorganic turbidity. The resulting formula is as follows:

chl = 0.3 x TP x (1 - exp (ln(2) x -(4 x E)/Z))

Vollenweider [26,27,83] developed his models on the conceptual basis that the phosphorus concentration is determined by the loading rate and modified by the sedimentation and the outflows. The following assumptions are made in the Vollenweider models:

1. The loading rate of phosphorus, the inflows and outflows, and the sedimentation rates remain constant during the year.

2. The lake is considered as one completely mixed compartment.

3. The phosphorus concentration in the inflow is the same as in the lake.

4. Sedimentation is proportional to the concentration for phosphorus.

5. Phosphorous is the limiting nutrient.

With these assumptions the solution of the steady-state equation becomes:

TP = L/(Z x (s + r))

where:

TP = concentration of total phosphorus in the lake in mg/m

L = annual load of P in mg/m^2
Z = mean depth of the lake in meters
s = sedimentation rate of phosphorus per year
r = turnover rate of the water (year-1).

The inclusion of the sedimentation term is essential, but it introduces a difficulty because the sedimentation rate is difficult to measure. Vollenweider has proposed a solution to the problem by using a statistical value for the sedimentation rate obtained from data for a large number of lakes. The final formula then becomes:

$$TP\ (predicted) = (L/q)\ /\ (1 + V\ (Z/q))$$

where q is the areal water load in m/y.

One may note that Z/q is the residence time of the water in the lake, Tw, measured in years. The key term is the sedimentation rate of P (s in equation above), and the residence time of the phosphorus (Tp) depends on the sedimentation rate and the residence time as follows:

$$T_p = (1/(1/Tw + s))$$

Statistical evaluations using data from a number of lakes in cold and temperate climates have yielded the following estimates for the phosphorus sedimentation rate:

s = (10/Z) (per year) [27]
s = (12.4/Z) (per year) [86]

Castagnino [6] has shown conclusively that these estimates are not necessarily valid in warm climates. This author suggests a higher value for the sedimentation rate in warm lakes:

s = (30/Z) (per year)

The work by Castagnino shows the importance of careful studies of the premises for each particular case before applying empirical models to certain groups of lakes. Mean annual concentrations of chlorophyll for different groups of lakes have been calculated using predicted total phosphorus concentrations and regression equations (Table 10).

The mean Secchi disc depth and the oxygen depletion rate in the hypolimnion can be calculated in a similar fashion using the total areal phosphorus load as the independent parameter [4].

Table 10. *Mean annual concentration of chlorophyll in selected groups of lakes*

Lake group	Chlorophyll concentration	N	r	Reference
USA	$0.39 \times (\text{TP pred})^{0.79}$	20	0.89	[87]
Nordic	$0.13 \times (\text{TP pred})^{1.03}$	13	0.82	[88]
Alpine	$0.47 \times (\text{TP pred})^{0.78}$	12	0.94	[89]
OECD	$0.37 \times (\text{TP pred})^{0.79}$	67	0.88	[3]

Several modifications have been suggested to the basic Vollenweider models. Imboden & Gächter [90] have developed a model in which different fractions of phosphorus and algal biomass are described as continuous functions of time and depth. This model takes into account the morphology of the lake, the hydraulic load, the respiration rate, the sedimentation rate, the vertical eddy diffusion, the depth of the thermocline and the exchange of phosphorus between the sediments and the water mass. An important improvement of the Vollenweider models has been suggested by Chapra & Reckhow [91] and Ciecka et al. [92]. The procedure involves the evaluation of the variance of the predictions, and it assures that the dispersion in the data is not neglected when the model predictions are evaluated.

6.3 Simulation models

In the present context, the term simulation model refers to a mathematical model describing phenomena that are dynamically determined by several processes. One such model, developed by Thomann in 1963 [93], describes the evolution of the concentration of dissolved oxygen in an aquatic system. The dissolved oxygen is mainly determined by the photosynthesis, the respiration rate and the equilibrium with the atmosphere. Another example is the model describing phosphorus concentrations in eutrophied lakes developed by Lung et al. [94].

However, the most important examples of simulation models are probably the models describing the thermal balance and the development of the thermocline in lakes in all climatic regions. Such simulations are extremely important for deep lakes and reservoirs since an understanding of the stratification and mixing processes is essential for making forecasts of the response of a lake or a reservoir to, for example, variations in the phosphorus load. Many models of this type have been developed over the years; here we mention only the heat balance model developed by Liljeblad [95,96], which have been used during a study of eutrophication in reservoirs in north Africa. The first version of this model was developed for the Sidi Mohammed Ben Abdellah reservoir on the Bou Regreg River in Morocco. The model incorporates descriptions of both the dispersion factors and, in a very approximate fashion, advective currents.

6.4 Ecological models

In contradistinction to the models discussed above, these models include mathematical formulations for a large number of physical, chemical and biological processes in a water body. The concept of ecological models originates from the work by Riley [82,97] and Steele [98]. Ecological models have attained their modern form through the work by Chen [99] and di Toro et al. [100]. Recent examples of ecological models are the models developed by Thomann et al. [101] and by Scavia [102]. Recent dynamic eutrophication models usually incorporate three types of term:

- hydrological and hydrodynamic terms;
- terms describing chemical and biological transformations;
- terms describing external loads and losses, both internal and external.

An ecological model based on the work by Scavia [70], Schnorr [103], di Toro et al. [100] and Schwartzman [104] also has been developed by the Institute of Water Quality in Denmark [105]. This model, which has been used in several eutrophication studies of Danish lakes, has been rewritten and extensively modified for the case of the warm water reservoir Sidi Mohammed Ben Abdellah in Morocco [106] and applied on several reservoirs in the north African region [107]. In its present form this model, which also has been introduced on microcomputors, comprises 15 state variables in a number of horizontal compartments (usually 5-6). In the application on the Sidi Mohammed Ben Abdellah reservoir, the water body was divided into five segments (in the vertical sense), which yielded a total of 75 steady-state variables which were integrated simultaneously. The data for the mass exchange between the segments were obtained from simulations with the heat balance model developed by Liljeblad (see above). The model calculates, among other things, the oxygen concentrations in the epilimnion and the hypolimnion at the end of the stratification period, the biomass, the primary production and the amount of detritus sedimented towards the hypolimnion. A more detailed description of the ecological model used during the eutrophication studies in north Africa is given in the second part of this report.

7. POSSIBLE STRATEGIES FOR EUTROPHICATION CONTROL IN LAKES AND RESERVOIRS

7.1 General considerations

After the real or probable causes for eutrophication have been identified, several possibilities usually exist for improving the existing situation, that is, to reverse or stop the degradation of the water quality. Basically, two strategies are possible: one may seek to eliminate or at least control the causes for the problem or one may simply fight the negative symptoms. In the first case, the aim is to control the primary production, for example, by reducing the nutrient load or by hindering the recycling of the nutrients in the water body. In the second case, the aim is to design methods for manipulating the produced biomass or to reduce the nuisance resulting from a strong oxygen consumption in the water, that is, the deficits of oxygen and its consequences. Methods of this kind are in general effective only during limited time periods and must, consequently, be repeated at regular intervals.

Eutrophication control methods can be classified in several ways; for example, one may distinguish between measures taken in the drainage basin and methods applicable in the water body proper. In certain cases, efficient measures at the water treatment level may be designed.

A classification which has been used by several authors is given below:

- physical methods, that is, manipulating or modifying the hydraulic and/or thermic situation in the reservoir;

- chemical methods by which, for example, the chemical form of the nutrients or the oxidizing agents are modified;

- biological methods by which the trophic chain in the lake/reservoir is modified.

Based on the objectives of the specific efforts, another way of classifying the possible actions is as follows:

- methods aimed at reducing the external load of nutrients;

- methods aimed at reducing the mobilization of nutrients from the sediments and the ensuing nutrient transport towards the epilimnion;

- methods aimed at preventing or retarding the multiplication of the algae;

- methods aimed at reducing or eliminating the organic matter produced by photosynthesis from reaching the hypolimnion.

- methods aimed at increasing the concentration of dissolved oxygen or of other oxidizing agents in the hypolimnion;

- methods aimed at manipulating the volume of the hypolimnion or the duration of the stagnant period.

Several reviews on available techniques for the rehabilitation of eutrophied lakes and reservoirs have appeared in the literature [cf. 4,5,108].

A short review of eutrophication control techniques is presented below where a distinction is made between methods relating to the drainage basin and methods applicable to the water mass of a lake or a reservoir.

7.2 Methods applicable in the drainage basin

These methods serve above all to reduce the external load of nutrients, and sometimes also of the external load of oxygen-consuming organic matter, to a lake or a reservoir. Two possible situations are possible concerning the nutrient sources in the drainage basin, which give rise to entirely different actions:

- one or several important nutrient point sources can be identified;

- the sources are mainly diffuse, that is, the nutrient load originates from sources such as agriculture or natural erosion.

7.2.1 Control of pollution point sources

In this case, the sources may be, for example, untreated sewage water from municipalities, yeast factories, sugar factories or other industries and aquaculture plants. A large number of practical methods exist; in general, they are based on chemical precipitation for eliminating or reducing the phosphate content in municipal and industrial effluents. In warm countries, water treatment methods can sometimes be designed on the basis of biological schemes in which different aquatic plants (e.g. water hyacinth) that have a high capacity for fixating the phosphorus are used instead of chemical agents.

A measure that sometimes can be effective is to introduce improved practices in polluting industries, for example, a more efficient collection of dung in cattle-breeding plants or modifications of industrial processes which aim to reduce the phosphorus (or, if needed, nitrogen) losses. In the case of residual water from towns and municipalities, restrictions may be imposed on the usage of phosphate-rich detergents.

The extent of the effort and the appropriate measures should never be decided before a careful analysis of the problem has been carried out. Among other things, the analysis must provide information on the relative importance of each pollution source as far as the eutrophication of the reservoir is concerned, and assign a value to the protection of the water body.

7.2.2 Control of nutrient losses from soil in drainage basin

In semi-arid and arid regions, which usually do not possess a protective vegetation cover and where the yearly precipitation usually arrives during a short period, the erosion contribution to the diffuse nutrient load is often substantial. These nutrient loads can be reduced in a number of ways:

- foresting certain parts of the drainage basin, in particular steep slopes;

- establishing a soil use programme in agricultural areas;

- restricting fertilizer use in the area;

- controlling the cattle pressure;

- planting "tree curtains" close to the rivers.

All these measures aim at better managing the terrain in the drainage basin and constitute a part of a global and rational planning of the whole lake/reservoir-drainage basin system.

7.2.3 Effluent treatment

In cases where the diffuse sources dominate the nutrient load to a lake or reservoir, an improved soil use policy may not be possible because, for example, social or economic constraints prevent an efficient implementation of new practices. In such situations, it may become necessary to consider solutions aimed at a direct reduction of the nutrients in the effluents before they enter the the water body, such as:

- construction of prereservoirs (a type of bioreactor), where the nutrients are captured in such a way that the nutrient load on the principal reservoir is reduced;

- affluent treatment by physical and chemical methods, for example, by filtration and/or flocculation or by direct injection in the affluents of chemicals which precipitate the phosphate;

- deviation of the most nutrient-rich affluent (e.g. by canalization).

7.3 Methods applicable in reservoirs

After having discussed methods aimed at treating the causes of eutrophication (i.e. reductions of the nutrient loads), we will now turn to methods designed mainly for reducing the symptoms of eutrophication.

7.3.1 Modifications of the hydraulic situation

Examples of methods aimed at modifying the hydraulic situation in a water body are listed below:

1. Shallow lakes may be made deeper, for example, by dredging the sediments or by increasing the height of the dam. The principal objective of this measure is either to diminish the macrophyte production in the shallow parts of the lake or to create a sufficiently deep lake for establishing a thermal stratification. The establishment of a thermocline prevents a direct contact between the nutrient-rich bottom waters and the trophogenic zone.

2. Nutrient-rich water is replaced with nutrient-poor water. This may be attained either by restituting the water through pumping (for lakes) or by opening the bottom gate (for reservoirs) followed by an automatic refill by groundwater or the adduction of external water of a good quality. To improve the efficiency of the process, the restitution may be done selectively (e.g. by a siphonage at the end of the stratification period when the bottom water is particularly poor in dissolved oxygen and rich in phosphorus and ammonium).

3. The stratification period in monomictic lakes and reservoirs may be shortened. This can be accomplished either partially, during a limited period, or through a complete destruction of the thermocline. The most efficient method is generally to advance the mixing in the autumn through an injection of compressed air at the bottom of the lake or by pumping surface water towards the hypolimnion. The advantage of initiating an early mixing of the water mass in the autumn is that a more efficient oxygen transport is generated in the water mass serving to oxidize hydrogen sulfide and also to oxidize the sediment surface that prevents releases of phosphate to the water body. Inducing a complete circulation of the water for the whole normally stratified period may have several disadvantages: an extended contact between the algae and the nutrients, which prolongs the production period, and an increase in temperature in the whole water mass to uncomfortable levels.

4. The water in the epilimnion is circulated during the early part of the production period, which in general coincides with the period when the stratification begins to form. This method may be efficient in lakes and reservoirs where the heating of the surface water is rapid and where the thermocline is formed close to the

surface. The effect of the circulation is to lower the level of the thermocline and thus to enhance a transport of algae out of the euphotic zone. This measure leads to both a decreased primary production and a weakening of the stratification with a resulting increased oxygen transport across the thermocline.

7.3.2 Chemical methods for fixing nutrients in sediments

After a new reservoir has been filled with water, a certain quantity of nutrients is mobilized as a result of the decomposition of the submerged vegetation. This nutrient mobilization is reduced by the natural sedimentation in the reservoir. To eliminate the available nutrients, in particular phosphates transported to the reservoir from the exterior, a chemical treatment of the reservoir water may be possible in certain cases. This entails the addition of chemicals that form stable insoluble compounds with the phosphate, such as aluminum, calcium or iron salts. In this way, the available phosphate is inactivated or entirely withdrawn from the primary production cycle: sedimentation of algae, decomposition, remobilization. The obvious disadvantages are the toxic impact of the flocculants and interference with the benthic fauna.

7.3.3 Harvesting of aquatic plants

This method is in particular applicable in cases where the negative effect of the eutrophication is a proliferation of macrophytes. The macrophytes may either be growing from the bottoms of shallow lakes or be floating species such as *Eichornia* or *Salvinia*. The idea is simply to remove mechanically the biomass of the plants before the decomposition takes place and thus to obtain a reduction in nutrients fixed in the plants and to avoid anoxic conditions in the water resulting from the decomposition of the plants. The disadvantage is that the conditions for phytoplankton production often are drastically improved when the macrophytes are removed.

7.3.4 Biological methods

As indicated, biological methods designed to improve the water quality in a eutrophied reservoir are difficult to apply in a proper fashion. One necessary prerequisite for applying this kind of method is a good knowledge of the structure and the dynamics of the ecosystem in question. Without such knowledge, identification of the elements responsible for the disequilibrium of the system cannot be ensured. It is also necessary to be able to foresee secondary consequences on other levels in the system than the ones considered as targets for a biological intervention.

Biological solutions for eutrophication control are in general most interesting to apply to young ecosystems (i.e. reservoirs). Biological methods are usually designed to reduce the algal biomass in order to diminish the amount of organic material which sediments towards the hypolimnion and thus the consumption of oxygen and nitrate in the deep parts of the reservoir. One may again note that a young ecosystem in a reservoir usually is far from maturity and thus displays disequilibrium symptoms, such as eutrophication and an ensuing degradation of the water quality. Biological methods should thus be designed to help the system to attain a state of equilibrium in a shorter time than if the system is left alone. These facts illustrate the difficulties in applying biological methods since they demand a very deep understanding of the ecosystem. If these methods are well chosen and carefully applied, they are, on the other hand, reassuring as they represent ecological solutions adapted to the ecosystem in question.

The fundamental disequilibrium often displayed by young reservoirs consists basically of an underrepresentation of the trophic level of primary consumers, that is, the consumers of phytoplankton form a population that is too small and too variable in time to control the algal growth efficiently. Primary consumers are mostly zooplankton (in particular cladoceres) and sometimes algivorous fish. Of course, the zooplankton population also is controlled by predatory fish which form the secondary consumer level.

An equilibrated ecosystem has the means for controlling the secondary consumers (e.g. by introducing large carnivorous fish species). In young ecosystems, however, the efficiency of the fish species present in the reservoir very likely is low and the introduction of carnivorous fish species on the third consumer level in such cases is an inefficient control method.

Three types of method can be envisioned to manipulate young ecosystems in order to equilibrate the system and to diminish the negative consequences of an exaggerated algal growth: introduction of algivorous fish species, intensive fishing of fish species which consume zooplankton and introduction of large carnivorous fish species.

(a) Introduction of algivorous fish species

The selection of the algivorous fish species must not be done until a serious analysis of the problem has been carried out. One must, for example, know which algal species cause(s) most of the problem and during which periods of the year this occurs. One must identify algivorous fish species which are adapted to the prevalent conditions in the reservoir; the selected species must be an efficient consumer of the predominant algae and no large carnivorous fish species should be present in the reservoir which would deplete rapidly the introduced algivores. One must also consider if the introduced species should not reproduce in the reservoir (in this case one can terminate the experiment if the negative effects of the measure are found to dominate over the positive ones) or if one wants to introduce the new species permanently into the system.

A certain experience has been gained for the following species in the context of eutrophication control:

1. The species *Mugil* (mullet). This species is particularly efficient in consuming algal species with large cells, such as *Peridinium* and *Ceratium* or species forming colonies or filaments (Cyanophyseae). Normally the mullets reproduce in the sea and not in sweet water; after spawning, the fry reenter the rivers to grow in sweet water. Mullets are cultivated in many countries.

2. The species *Tilapia*. Among the many *Tilapia* species, *T. machrochir* is considered to be the most specialized in terms of phytoplankton consumption. This species also consumes algal cells of a considerable size. The *Tilapia* reproduces easily and quickly in sweet water. However, the fast reproduction represents the most serious problem with the *Tilapia* because it makes population control extremely difficult. Moreover, the size of the individuals tends to decrease as the population becomes large. As a result, a surplus of fish of a very small size may be produced that are neither efficient consumers of algae nor desirable for commercial fishing.

3. The silver carp (*Hypophthalamichtys molitrix*). Of Chinese origin, this species has been used in a large number of countries to control algal biomass in lakes

and reservoirs. The species is very interesting because of its capability to consume smaller algae (algae with cell diameters between 10 and 50 um) than any other fish species. It is very improbable that the silver carp would reproduce in a reservoir since it needs very special conditions for its reproduction. This is in fact an advantage if one wants to control the population. Artificial reproduction of the silver carp is well known and practised in many laboratories and aquaculture institutes. If the silver carp is introduced in a reservoir, its population must be reinforced from time to time since the large specimens will be caught occasionally.

(b) Intensive fishing of fish species that consume zooplankton

Before launching an intensive fishing campaign of secondary consumers in order to decrease the grazing pressure on the zooplankton population, the structure and dynamics of the food chain during all the entire year must be fully known. It is at any rate difficult to reach an equilibrated ecosystem with a built-in algal control with this procedure.

(c) Introduction of large carnivorous fish species

The justification for introducing large carnivorous fish species in a eutrophied reservoir is to reinforce the trophic level serving to control the secondary consumers. Evaluating the effects of such a measure is even more difficult than evaluating the effects of intensive fishing campaigns. Also, an introduction of a large carnivorous fish species is incompatible with an introduction of algivorous species.

7.3.5 Measures for increasing reserves of oxidizing material

The goal of several of the methods modifying the hydraulic situation in a reservoir is in fact to increase the reserves of dissolved oxygen in the hypolimnion. For example, an increase of the volume of the hypolimnion or a shortening of the stratified period serves in principle to increase the quantity of dissolved oxygen available for decomposition and oxidation of organic material. The expected result of these measures is that oxygen-deficient situations and anoxic conditions in the hypolimnion, with all its consequences, should be attained later in the year or not at all.

In certain cases, when the eutrophication already is very advanced, the depletion rate of oxygen in the hypolimnion may be very high. In such situations, more radical measures may be used with the only purpose to reestablish aerobic conditions in the hypolimnion:

1. Injection of compressed air in the hypolimnion without destroying the thermal stratification. Several techniques have been developed to attain this goal. The principle is to aerate water pumped up from the hypolimnion and then to reintroduce the now-oxygenated water into the deep water layers in the reservoir.

2. Injection of nitrate into the hypolimnion. Dissolved nitrate serves as an oxidizing agent in the hypolimnion. When almost all the dissolved oxygen has been depleted, the denitrification starts to provide oxygen to the decomposition processes. Not until all the dissolved oxygen and all the nitrate have been consumed in the hypolimnion does sulfate start to serve as an oxidizing agent. During this phase of the decomposition process, hydrogen sulfide is being formed. To avoid this situation, during which the water quality severely deteriorates, one may increase the stock of nitrates in the hypolimnion by a direct injection of calcium nitrate. It is important to make sure that all the nitrate is injected into the hypolimnion, well below the trophogenic zone; otherwise the nitrate may become used for the algal reproduction.

REFERENCES

1. Vollenweider, R.A. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors in eutrophication. Paris, Organisation for Economic Co-operation and Development, 1968 (OECD Technical Report DAS/CSI/68.27).

2. Eutrophication. Proceedings of the 10th Nordic Symposium on Water Research. Olso, Nordic Secretariat of Environmental Science, 1975 (in Scandinavian languages).

3. Eutrophisation des eaux. Méthodes de surveillance, d'évaluation et de lutte. [Eutrophication of water. Methods of surveillance, evaluation and control] Paris, Organisation of Economic Co-Operation and Development, 1982.

4. The control of eutrophication of lakes and reservoirs. Paris, United Nations Educational, Scientific and Cultural Organization, 1986.

5. Landner, L. Eutrophication of lakes. Causes, effects and means for control, with emphasis on lake rehabilitation. Copenhagen, WHO Regional Office for Europe, 1976 (unpublished document ICP/CEP 210).

6. Castagnino, W.A. Investigactión de modelos simplificados de eutroficación en lagos tropicales. [Investigation of simple models of eutrophication in tropical lakes]. Lima, Pan American Centre for Sanitary Engineering and Environmental Sciences, 1982.

7. Salas, H.J. Desarrollo de metodologias simplificadas para la evaluación de eutroficación en lagos calidos (antes lagos tropicales). [Development of simplified methodology for the evaluation of eutrophication in warm lakes (tropical lakes previously] Lima, Pan American Centre for Sanitary Engineering and Environmental Sciences, 1983.

8. Vollenweider, R.A. Aspects particuliers de l'eutrophisation des lacs-réservoirs en climat chaud. [Particular aspects of eutrophication of lakes and reservoirs in warm climates] Copenhagen, WHO Regional Office for Europe, 1982 (unpublished document MOR/RCE 001(2)/13).

9. Carmichael, W.W. et al. Algal toxins and water-based diseases. CRC critical reviews in environmental control, 15: 275-312 (1985).

10. Guidelines for drinking-water quality. Vol. 1, Geneva, World Health Organization, 1984.

11. Stewart, K.M. & Rohlich, G.A. Eutrophication - a review. Sacramento, CA, Water Quality Control Board of California, 1967 (Publication No. 34).

12. Eutrophication: causes, consequences, correctives. Washington, DC, National Academy of Sciences, 1969.

13. Lee, G.F, et al. Eutrophication of water bodies: Insights for an age-old problem. Environmental science and technology, 12: 900-908 (1978).

14. Welch, E.B. Ecological effects of waste water. Cambridge, Cambridge University Press, 1980.

15. Landner, L. et al. Protection de la qualité des eaux des lacs-réservoirs au Maroc. Rapport Final. [Protection of water quality of lakes and reservoirs in Morocco] Nairobi, United Nations Environment Programme, 1982 (unpublished document UNEP/MOR/79/005).

16. Proceedings of the Eutrophication Workshop. Canberra, Australian Water Resources Council, 1983 (Conference Series No. 7).

17. Thornton, J.A. Aspects of eutrophication management in tropical/subtropical regions: A review. Presented at the IWPA meeting, 1983, Amsterdam. Amsterdam, IWPA, 1983.

18. Weber, C.A. Aufbau und Vegetation der Moore norddeutschlands. [Structure and vegetation of marshy soils in southern Germany] Botanische Jahrbücher für Systematik, Pflanzengeschichte und Pflanzengeographie, 40: 19-31 (1907).

19. Naumann, E. [Some views concerning the ecology of limnoplankton]. With special reference to phytoplankton. Svensk botanisk tidskrift, 13: 129-163 (1919) (in Swedish).

20. Thienemann, A. Untersuchungen über die Beziehungen zwischen dem Sauerstoffgehalt des Wassers und der Zusammensetzung des Fauna in norddeutschen Seen. [Studies on the relation between the amounts of oxygen in the water and the composition of the fauna in the lakes of northern Germany] Archiv für hydrobiologie, 12: 1-65 (1918).

21. Welsh, P.S. Limnology, 2nd ed. New York, McGraw-Hill, 1952.

22. Sawyer, C.N. Basic concepts of eutrophication. Water Pollution Control Federation journal, 38: 1237-1258 (1966).

24. Lee, G.F. Eutrophication. In: Encyclopedia of chemical technology, supplemental volume. New York, John Wiley & Sons, 1971.

25. Golterman, H.L. Methods for physical and chemical analysis of fresh waters, 2nd ed., Oxford, Blackwell Scientific Publications, 1978 (IBP Handbook No. 8).

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

27. Vollenweider, R.A. Advances in defining critical loading levels for phosphorus in eutrophication. Memorie Istituto Italiano di Idrobiologia, Dott Marco de Marchi, 33: 53-83 (1976).

28. Balon, E.K. Kaziba: the dubious benefits of large dams. Ambio, 7: 40-48 (1978).

29. Bernhardt, H. et al. Vergleichende Untersuchungen. Ermittlung der Eutrophierungsvorgänge und ihrer Ursachen an Riveris- und Wahnbachtalsperre. [Comparative investigation. Research on the process of eutrophication and its causes in the Riveris and Wahnbachtalsperre dams] Wasser, 40: 245-303 (1973).

30. Bernhardt, H. Pronostics sur la qualité de l'eau dans le barrage de Sidi Salem, Tunisie. Deuxième partie. [Prognosis of the quality of water in the Sidi Salem reservoir, Tunisia] Tunis, Ministry of Agriculture, 1981.

31. Nusch, E.A. Comparative investigations on extent, causes and effects of eutrophication in western German reservoirs. Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19: 1871-1879 (1975).

32. Steel, J.A. The management of Thames valley reservoirs. In: Proceedings of the Water Research Centre Symposium of the Effects of Storage in Water Quality. Medmenham, Water Research Centre, 1975.

33. Duncan A. The importance of zooplankton in the ecology of reservoirs. In: Proceedings of the Water Research Centre

Symposium on The Effects of Storage on Water Quality. Medmenham, Water Research Centre, 1975.

34. Burgis, M.J. Case studies of lake ecosystems at different latitudes of the tropics. The Lake George ecosystem. Internationale Vereinigung für Theoretische und Angewandte Limnologie, 18: 647-655 (1978).

35. Koeppen, W. Handbuch der Klimatologie [Handbook of climatology] Berlin, Borntraeger, 1931.

36. Serruya, C. & Pollingher, U. Lakes of the warm belt. Cambridge, Cambridge University Press, 1983.

37. Readle, L.C. The inland waters of tropical Africa. London, Longman, 1974.

38. Talling, J.F. The annual cycle of stratification and phytoplankton growth in Lake Victoria (East Africa). Internationale Revue der gesamten Hydrobiologie, 51: 545-621 (1966).

39. Rutter, F. Planktonstudien der Deutschen Limnologischen Sunda Expedition. [Studies on plankton of the German limnologic Sunda expedition] Archiv für Hydrobiologie, Supplement 21: 1-274 (1952).

40. Lewis, W.M. Jr. The thermal regime of Lake Lanao (Philippines) and its theoretical implications for tropical lakes. Limnology and oceanography, 18: 200-217 (1973).

41. Hutchinson, G.E. A treatise on limnology. I. Geography, physics and chemistry. New York, John Wiley & Sons, 1957.

42. Richerson, P.J. et al. A survey of the physical and chemical limmology of Lake Titicaca. Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19: 1498-1503 (1975).

43. Stanhill, G. & Neumann, J. Energy balance and evaporation. In: Serruya, C., ed. Lake Kinneret. Hague, Junk Publishers, 1978 (Monographiae Biologicae No. 32).

44. Ruttner, F. Hydrographische und hydrochemische Beobachtungen auf Java, Sumatra und Bali. [Hydrographical

and hydrochemical observations of Java, Sumatra and Bali] Archiv für Hydrobiologie, Supplement 8: 197 454 (1931).

45. Ruttner, F. Stabilität und Umschichten in tropischen und temperierten Seen. [Stability and changes in stratification in tropical and temperate lakes] Archiv für Hydrobiologie, Supplement 15: 178-186 (1937).

46. Baxter, R.M. et al. Stratification in tropical African lakes at moderate latitudes (1500-2000 m). Limnology and oceanography, 10: 510-520 (1965).

47. Talling, J.F. The photosynthetic activity of phytoplankton in East African lakes. Internationale Revue der gesamten Hydrobiologie, 50: 1-31 (1965).

48. Lewis, W.M., Jr. Primary production in the plankton community of a tropical lake. Ecology monographs, 44: 377-409 (1974).

49. Serruya, C., et al. Lake Kinneret sediments: nutrient composition of the pore water and mud-water exchanges. Limnology and oceanography, 19: 489-508 (1974).

50. Gliwicz, Z.M. Metalimnetic gradient and phosphorus loss from the epilimnion in stratified eutrophic lakes. In: Barica, J. & Mur, L.R., ed. Hypertrophic ecosystems. Hague, Junk Publishers, 1980.

51. Cullen, P. & Smalls, I. Eutrophication in semi-arid areas - the Australian experience. Water quality bulletin, 6: 79-90 (1981).

52. Rosich, R.S. & Cullen, P. Sediments, algae, nutrients - interrelationships in Lakes Burley, Griffin and Ginninderra. Internationale Vereinigung für Theoretische und Angewandte Limnologie, 21: 1009-1016 (1981).

53. Serruya, C. et al. Nitrogen and phosphorus balances and load-biomass relationship in Lake Kinneret (Israel). Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19: 1357-1369 (1975).

54. Landner, L. et al. Protection de la qualité des eaux des lacs-réservoirs au Maroc. Rapport intérimaire. [Protection of the water quality in reservoirs of Morocco.

Interim report]. Nairobi, United Nations Environment Programme, 1980 (UNEP/MOR/70/005).

55. Falkowski, T. La qualité de l'eau des oueds alimentant la retenue du barrage Sidi Mohammed Ben Abdellah, juillet 1979-décembre 1980. [Water quality of water sources flowing into the Sidi Mohammed Ben Abdellah reservoir, July 1979-December 1980] Nairobi, United Nations Environment Programme, 1982 (Technical Report, Series A, No. 18).

56. Bouloud, A. & Lyden, A. Production primaire et nombre des phyto- et zooplanctons dans le réservoir Sidi Mohammed Ben Abdellah, 1976 - mai 1980. [Primary production and number of phyto- and zooplankton in the Sidi Mohammed Ben Abdellah reservoir, 1976-May 1989] Nairobi, United Nations Environment Programme, 1980 (Technical Report, Series A, No. 10).

57. Bourich, L. Phyto- et zooplanctons, biomasse et production primaire dans le réservoir Sidi Mohammed Ben Abdellah, avril 1980-décembre 1981. [Phyto- and zooplankton, biomass and primary production in the Sidi Mohammed Ben Abdelleh reservoir, April 1980-December 1981] Nairobi, United Nations Environment Programme, 1982 (Technical Report, Series A, No. 10).

58. Saadoui, M. Eutrophisation: Rapport intérimaire sur la qualité de l'eau de la retenue de Sidi Salem, Tunisie. [Eutrophication. Interim report on the water quality of the reservoir of Sidi Salem, Tunisia] Tunis. Ministry of Agriculture, Department of Water Resources, 1986.

59. Pollingher, U. The phytoplankton of Lake Kinneret. In: Serruya, C., ed. Lake Kinneret. Hague, Junk Publishers, 1978 (Monographiae Biologicae No. 32).

60. Wetzel, R.G. Variations in productivity of Goos and hypertrophic Sylvan Lakes, Indiana. Investigation of Indiana lakes and streams, 3: 147-184 (1966).

61. Richerson, P.J. er al. The limnology of Lake Titicaca (Peru - Bolivia), a large, high-altitude tropical lake. Davis, University of California, 1977 (Institute of Ecology Publication No. 14).

62. Megard, R.O. Phytoplankton, photosynthesis and phosphorus in Lake Minnetonka, Minnesota. Limnology and oceanography, 17: 68-87 (1972).

63. Hecky, R.E. & Fee, E.J. Primary production and rates of algal growth in Lake Tanganyika. Limnology and oceanography, 26: 532-547 (1963).

64. Goldman, C.R. & Wetzel, R.G. A study of the primary productivity of Clear Lake, Clear County, California. Ecology, 44: 283-294 (1963).

65. Vollenweider, R.A. et al. A comparative review pf phytoplankton and primary production in the Laurentian Great Lakes. Journal of Fishery Research Board Canada, 1: 739-762 (1974).

66. Moskalenko, B.K. Biological productivity system of Lake Baikal. Internationale Vereinigung für Theoretische und Angewandte Limnologie, 18: 568-573 (1972).

67. Rodhe, W. Primärproduktion und Seetypen. [Primary production and lake type] Internationale Vereinigung für Theoretische und Angewandte Limnologie, 13: 121-141 (1958).

68. Peterson, B.J. Radiocarbon uptake: Its relation to net particular carbon production. Limnology and oceanography, 23: 179-184 (1978).

69. Hecky, R.E. & King, H. The phytoplankton and protozooplankton of Lake Tanganyika: Species composition, biomass, chlorophyll content, and spatio-temporal distribution. Limnology and oceanography, 26: 548-564 (1971).

70. Scavia, D. Examination of phosphorus cycling and control of phytoplankton dynamics in Lake Ontario with an ecological model. Journal of Fishery Research Board Canada, 36: 1336-1346 (1979).

71. Steemann Nielsen, E. & Jörgensen, E.G. The adaptation of plankton algae. 1. General part. Physiology of plants, 19: 232-240 (1968).

72. Morris, I. & Farrell, K. Photosynthetic rates, gross patterns of carbon dioxide assimilation and activities of

ribulose diphosphate carboxylase in marine algae grown at different temperatures. Physiology of plants, 25: 372-377 (1971).

73. Rhee, G.Y. & Gotham, I.J. The effect of environmental factors on phytoplankton growth: Temperature and the interactions of temperature with nutrient limitation. Limnology and oceanography, 26: 635-648 (1981).

74. Abouzaid, H. & Landner, L. L'eutrophisation de lac-réservoir Sidi Mohammed Ben Abdellah près de Rabat, Maroc. [Eutrophication of Sidi Mohammed Ben Abdellah reservoir near Rabat, Morocco] Aquaculture, 6: 345-347 (1981).

75. Falkowski, T. La qualité de l'eau des oueds et les apports solides au réservoir Sidi Mohammed Ben Abdellah, octobre 1978 - juin 1979. [Water quality of the source waters and suspended matter entering the Sidi Mohammed Ben Adellah reservoir, October 1978-June 1979] Nairobi, United Nations Environment Programme, 1979 (Technical Report, Series A. No. 5 of Project MOR/RCE/001).

76. Pollingher, U. & Berman, T. Temporal and spatial patterns of dinoflagellate blooms in Lake Kinneret, Israel (1969-1974). Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19: 1370-1382 (1975).

77. Spataru, P. The feeding habits of *Tilapia galilea* (Artedi) in Lake Kinneret (Israel). Aquaculture, 9: 47-59 (1976).

78. Bouloud, A. Essais d'adaptation du mulet (Mugli spp). [Experiments in the adaptation of mullet (*Mugli* spp)] Nairobi, United Nations Environment Programme, 1979 (Technical Report Series A. No. 16).

79. Sherr, B.F. et al. Decomposition of organic detritus. A selective role for microflagellate Protozoa. Limnology and oceanography, 27: 765-769 (1982).

80. Scott, W.E. et al. Hartebeespoort Dam: A case study of a hypertrophic, warm, monomictic impoundment. In: Barica, J. & Mur, L.R., ed. Hypertrophic ecosystems. Hague, Junk Publishers, 1980.

81. Rosich, R.S. Lake modelling. In: Proceedings of the Eutrophication Workshop. Canberra, Australian Water Resources Council, 1983 (Conference Series No. 7).

82. Riley, G.A. Factors controlling phytoplankton populations on George's Bank. Journal of marine research, 6: 54-73 (1946).

83. Vollenweider, R.A. Möglichkeiten und Grenzen elementarer Modelle der Stoffbilans von Seen. [Possibilities and limitations of sample models on the balance of substances in lakes] Archiv für Hydrobiologie, 66: 1-36 (1969).

84. Sakamoto, M. Primary production by phytoplankton community in some Japanese lakes, and its dependence on lake depth. Archiv für Hydrobiologie, 62 1-28 (1966).

85. Verduin, J. et al. A simple equation relating total phosphorus to chlorophyll concentration in lakes. Internationale Vereinigung für Theoretische und Angewandte Limmologie, 10: 352 (1978).

86. Chapra, S.C. & Tarapckak, S.J.. A chlorophyll *a* model and its relationship to phosphorous loading plots for lakes. Water resources research, 12: 1260-1264 (1976).

87. Rast, W. & Lee, G.F. Summary analysis of the North American (U.S. portion) OECD eutrophication project: nutrient loading - lake response relationships and trophic state indices. Corvallis, OR, US Environmental Protection Agency, 1978 (Ecology Research Series, No. EPA-600/3-78-008).

88. Ryding, S.-O. Monitoring of inland waters: OECD eutrophication programme - the Nordic project. Oslo, Nordic Secretariat of Environmental Sciences, 1980.

89. Fricker, H. OECD eutrophication programme. Regional project, alpine lakes. Bern, Swiss Federal Board for Environmental Protection, 1980.

90. Imboden, D.M. & Gächter, R. A dynamic model for trophic state prediction. Ecological modelling, 4: 77-98 (1978).

91. Chapra, S.C. & Reckhow, K.H. Expressing the phosphorus loading concept in probabilistic terms. Journal of Fishery

Research Board Canada, 6: 225-229 (1979).

92. Ciecka, J. et al. A statistical model for small lake water management. Water resources bulletin, 15: 1308-1330 (1979).

93. Thomann, R.V. Mathematical model for dissolved oxygen. Environmental engineering, 89: 1-30 (1963).

94. Lung, W.S. et al. Phosphorus models for eutrophic lakes. Water research, 10: 1101-1114 (1976).

95. Liljeblad, Aa. A mathematical model describing the thermal balance and the transport mechanisms in the reservoir on the Bou Regreg River, Morocco. Stockholm, IVL, 1981 (Technical Report Series D, No. 2).

96. Liljeblad, Aa. Hydrological and hydrodynamic description of the reservoir Sidi Mohammed Ben Abdellah, Morocco. Stockholm, IVL, 1982 (Technical Report Series D, No. 3).

97. Riley, G.A. Theory of food-chain relations in the ocean. In: Hill, M.N., ed. The sea. New York, Interscience, 1963.

98. Steele, R.H. Plant production in Fladen Ground. Journal of the Marine Biology Association (UK), 35: 1-33 (1956).

99. Chen, C.W. Concepts and utilities of ecological models. Environmental engineering, 96: 1085-1097 (1970.

100. di Toro, D.M. et al. A dynamic model of phytoplankton populations in natural waters. In: Nonequilibrium systems in natural water chemistry. Washington, DC, American Chemical Society, 1971 (Advances in Chemistry Series, 106).

101. Thomann, R.V. et al. Estimated responses of Lake Ontario phytoplankton biomass to varying nutrient levels. Journal of Great Lakes research, 3 123-131 (1977).

102. Scavia, D. An ecological model of Lake Ontario. Ecological modelling, 8: 49-78 (1980).

103. Schnorr, J.L. & di Toro, D.M. Differential phytoplankton sinking and growth rates: An Eigen value analysis. Ecological modelling, 9: 233-245 (1979).

104. Swartzman, G.L. & Bentley, R. A review of plankton simulation models. ISBM journal, 1: 30-81 (1979).

105. Nielsen, P.V. A preliminary water quality model for the reservoir of Bou Regreg, Morocco. Horsholm, Danish Institute of Water Quality, 1980 (Technical Report Series D, No. 1).

106. Wahlgren, U. Modéle mathématique de la qualité de l'eau: Calibration et simulations des conséquences de diverses mésures pour combattre l'eutrophisation et de la surélévation de la retenue du barrage Sidi Mohammed Ben Abdellah, sur le Bou Regreg, Maroc. [Mathematical model of water quality: calibration and simulations of the consequences of different measures for combatting eutrophication and the raising of the levels of the Sidi Mohammed Ben Abdellah reservoir on the Bou Regreg, Morocco] Stockholm, IVL, 1982 (Technical Report Series D, No. 5).

107. Landner, L. & Wahlgren, U. Rapport final du sous-project "Controle de l'eutrophisation des lacs-réservoirs en climat chaud". Développement des ressources en eau en Afrique du Nord. [Final Report of project "Control of eutrophication of lakes and reservoirs in warm climates. Development of water resources in north Africa] Copenhagen, WHO Regional Office for Europe, 1986 (unpublished document ICP/CWS 050).

PART 2: CASE STUDIES

EUTROPHICATION OF SIDI MOHAMMED BEN ABDELLAH RESERVOIR IN MOROCCO

EVOLUTION OF THE TROPHIC STATE OF THE SIDI SALEM RESERVOIR IN TUNISIA

SIDI MOHAMMED BEN ABDELLAH RESERVOIR

1. INTRODUCTION

The Sidi Mohammed Ben Abdellah reservoir is situated on the Bou Regreg River about 20 km from Rabat, Morocco. The drainage basins of three rivers provide the water for the reservoir: Bou Regreb, Grou and Korifla. This important dam, which was constructed to provide domestic water to the four million inhabitants in the coastal zone between Casablanca and Rabat, was filled with water in 1974.

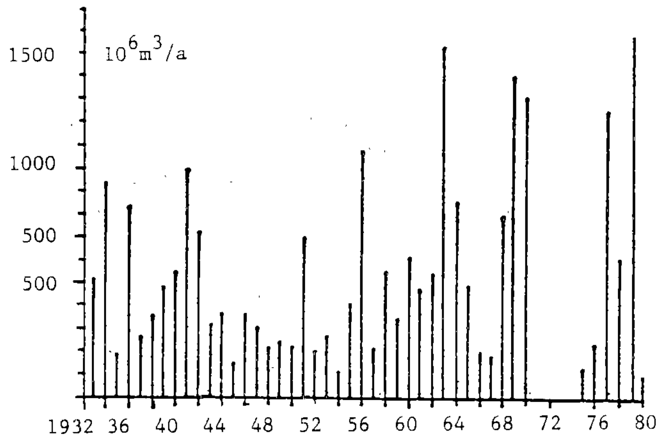
Water quality studies carried out by the National Office for Drinking Water (Office National de l'Eau Potable) showed an oxygen deficit in the reservoir by 1975, and all subsequent studies have shown a strong thermal stratification accompanied by a deoxygenation of the hypolimnic water for several months during the warm season. Hydrogen sulfide in the hypolimnion was found during the dry season in 1976-1981.

The degradation of the water quality has caused serious difficulties at the water treatment station at Bou Regreg. During the periods when the presence of hydrogen sulfide has made the hypolimnic water unusable, raw water has been drawn from the surface layers, which are very contaminated by algae. To offset the contamination, a water treatment station has had to add high levels of chlorine and aluminum sulfate and to use active carbon powder to decrease the bad odours. The situation has increased significantly the costs of drinking-water production without always attaining an entirely acceptable drinking-water quality.

The total annual flow varies considerably between years. Estimated flows for most of the years between 1932 and 1980 are shown in Figure 1. Also, the flows vary greatly for different periods during a year. The yearly inflows are concentrated in a few weeks during the winter (in particular during January and February). The rivers Grou and Korifla-Mechraa are usually dry from June to September. Bou Regreg usually has at least a small flow during the whole year.

Construction started in 1971 and was completed in 1974. The Sidi Mohammed Ben Abdellah reservoir was filled between 1973 and 1976; the normal level (50 m above mean sea level (AMSL)) was reached in April 1976. At the normal level, the volume of the reservoir is 500 Mm^3, the surface is 30 km^2 and the mean depth is 17 m (maximum depth is 35 m). The reservoir has four outlets situated between 2 and 27 m below the normal water level. The bottom gate, which is connected to a tunnel, has a

Fig. 1. *Total estimated inflows for 1932-1980*



capacity of 150 m^3/s, and the emergency release at 50 m AMSL has a capacity of 12 000 m^3/s.

2. STUDY OF LIQUID AND SOLID TRANSPORTS, 1974-1981

2.1 Transport of liquids

Table 1 gives the estimates by the Hydraulic Direction of

Table 1. *Total liquid transport to Sidi Mohammed Ben Abdellah reservoir, 1974-1981*

Year	Volume (Mm^3)	Mean annual flow (m^3/s)
1974-1975	164	5.2
1975-1976	270	8.6
1976-1977	1258	39.9
1977-1978	606	19.2
1978-1979	1599	50.7
1979-1980	425	13.5
1980-1981	157	5.0

the total liquid transport to the reservoir from the monthly hydraulic balances.

The inflows are calculated for each agricultural year, that is, September-August. The residence time in the reservoir varies between 0.3 years for the year 1978-1979 and 3.2 years for the year 1980-1981.

The annual distribution of the inflows is very uneven; inflows are concentrated during the flood periods in the winter. The largest flood during the study period occurred in February 1979 (about 1000 Mm^3), which meant that the water in the reservoir was theoretically exchanged twice during that month.

2.2 Transport of solids

2.2.1 Methods

The quality of the water in the three principal tributaries to the reservoir was investigated regularly with a sampling frequency of, in general, twice per month since October 1978. Exceptions have been flood periods, when the frequency was higher, and dry periods with very low or nonexistent flows, when the sampling frequency was decreased or even suspended.

Two floods, which occurred from 27 to 30 January 1979 and 18 to 20 January 1980, respectively, have been followed in detail with a sampling frequency of 3.5-4 h. They may be considered as medium to small flood, with flows between 60 and 230 m^3/s for the Bou Regreg River and Grou River, respectively, for the first flood and between 15 and 36 m^3/s for the second flood. Unfortunately, sampling could not be carried out during the important flood in February 1979. The samplings were made at the station Dar es Soltane (Bou Regreg), Ras El Fantia (Grou), and for the Korifla River and Mechraa River just above and below their confluence. The flows were estimated from river levels obtained at gauging stations in the vicinity of the sampling sites.

The water samples were analysed for total phosphorus, orthophosphate, the sum of nitrate and nitrite, Kjeldahl nitrogen, total suspended matter and total organic suspended matter. A knowledge of these parameters and the flows allows estimates of the total loss of suspended matter and nutrients to be made.

A study of the correlation between the concentration of suspended and dissolved matter and the measured flows also was carried out to find out if predictions of the total load of suspended matter or phosphorus from the flow distribution during the rainy season were justifiable [1].

To identify the regions in the drainage basin where the most important nutrient losses occurred, six sampling rounds were conducted between March 1978 and February 1980. Samples were obtained from 22 stations situated in different parts of the drainage basin. The rounds were designed to compare the situation during different flows.

Soil samples also were collected in September 1978 to determine the nutrient content.

2.2.2 Results and discussion

Table 2 shows the total liquid and solid inflow for 1978-1979 and 1979-1980, as well as the contributions from each sub-basin (except 1979-1980 for Korifla-Mechraa). The table also shows the fraction of the inflows that occurred during the floods (defined as flows superior to 10 m^3/s).

Clearly, almost all the solid material arrives with the floods during rainy winters.

Also, correlation studies show a linear log-log type relation between flow and suspended matter and total phosphorus for flows between 10 and 100 m^3/s. However, no correlation between flow and concentration for nitrate in the river water is seen.

The following conclusions were drawn about nutrient loss:

(a) Phosphorus and nitrogen loads are minor from these regions:

- the upper part of the Bou Regreg basin, in particular the Aguennour, Marrout and Ksiksou rivers;

- the upper basin of Grou, above the highway between Maaziz and Rommani;

- the sub-basin of the Korifla River.

(b) Nutrient loads are considerable from these regions:

- the sub-basin of the Tanouberte River;

- the lower basin of the Bou Regreg River (i.e. the region between Maaziz and the dam);

- the lower basin of the Grou River, the origin of the most important losses in nitrate and phosphorus in the entire drainage basin;

Table 2. *Solid and liquid inflow to Sidi Mohammed Ben Abdullah reservoir during 1978–1979 and 1979–1980*

Parameter	Sub-basin						
	Bou Regreg		Grou		Korifla-Mechraa	Total	
	1978–1979	1979–1980	1978–1979	1979–1980	1979–1978	1978–1979	1979–1980
Inflows							
Mm^3	880	230	624	195	96	1600	425
%	55	54	39	46	6	–	–
Suspended matter, kt	2200	430	1600	470	200	4000	900
%	54	48	39	52	7	–	–
% during floods	99.9	67	98.7	61	99.4	99.5	64
P tot, kt	2.4	0.14	2.7	0.25	0.2	5.3	0.39
%	46	36	51	64	3	–	–
% during floods	99.7	79	99.7	45	98	99.7	57
NO_3-N, kt	2.4	0.34	2.3	0.86	0.3	5	1.2
%	49	28	45	72	6	–	–
% during floods	94	46	83	56	65	86	53

- the sub-basin of the Mechraa River.

Furthermore, analyses of the soil samples show a remarkable uniformity in the phosphate and nitrate concentrations for the whole drainage basin. Thus, the most important factor determining nutrient losses is the vegetation cover.

2.2.3 Areal nutrient loads to the reservoir and areal nutrient losses in the drainage basin

The phosphorus loads to the reservoir are extremely high, in particular for the agricultural year 1978-1979. Even if the part of the influx that was immediately lost through the emergency release is subtracted for 1978-1979 (which accounts for 42% of the annual water inflow), the total phosphorus load was 3000 tons. This number is equivalent to an annual phosphorus surface load of 100 g/m^2.

If the well-known Vollenweider relations between phosphorus load and biological response are applied to forecast the degree of eutrophication for the Sidi Mohammed Ben Abdellah reservoir, the result falls almost outside the diagram because of the excessive phosphorus load. Even if the part of the phosphorus which is bound to mineral particles and thus inaccessible to the algae is corrected for, the predicted phosphate concentration in the water mass is very high. It should be noted that the fraction of orthophosphate in the rivers during floods is between 20 and 50% of the total phosphorus. Nevertheless, the eutrophication symptoms (e.g. chlorophyll concentrations, oxygen consumption in the hypolimnion) in the reservoir are far from the extreme levels which would be expected from the diagrams.

Why doesn't this enormous production potential correspond to the actual production in the reservoir? At least two explanations are possible. The first relates to biological factors in the reservoir proper; the floods transport large quantities of suspended matter to the reservoir, which in turn cause the productive zone to become very shallow because of the ensuing reduced light penetration in the water. The second explanation concerns the chemical processes in the inflowing water; the incoming phosphate may become inaccessible to the algae because of flocculation processes involving fine clay particles in the rivers and/or by a formation of hydroxyapatite in the reservoir. The latter hypothesis is in fact supported by the result of a series of analyses of orthophosphate in the surface and at the bottom of the reservoir. The very high phosphate concentrations in the epilimnion (170 ug/l) right after the flood fall rapidly at the same time as the phosphate concentration in the hypolimnion rises so as to make the available phosphate concentration in the epilimnion attain a value of around 15 ug/l a few weeks after the flood. For the agricultural year 1979-1980, which was a medium year regarding inflow (425 Mm^3 which corresponds to a mean annual flow of 13.5 m^3/s) and also a year without any major floods, the total phosphorus load was estimated at 10 g/m^2. By introducing the mean depth (16 m) and the retention time of the water (1.06 y) into the Vollenweider diagrams, one arrives at a prediction of a eutrophic or even hypertrophic state of the reservoir (see Fig. 2). It may be reasonable, given the above discussion, to replace the total phosphorus load with the total phosphate load and repeat the exercise. The phosphate fraction in the inflows at flows above 20 m^3/s is about 25%, which would lead to an annual phosphate surface load of 2.5 g/m^3, a result which again would lead to a classification of the reservoir as eutrophied with high chlorophyll concentrations and a hypolimnic oxygen consumption rate which is not observed in the reservoir. These results lead to the conclusion that our knowledge of the limnology of lakes and reservoirs in warm climates, including semi-arid areas, must be developed in order to be able to apply Vollenweider-type models to lakes in this climatic region.

The mean values (in kg/ha/y) of nutrient losses by erosion for the drainage basin of the reservoir were the following:

Year	Total P	$N\text{-}NO_3$
1978-1979	5.3	5
1979-1980	0.8	1-1.5

These results show that phosphorus losses are more important than those generally found in temperate regions: 0.06 kg/ha/y for areas covered by forest and 0.14-0.4 kg/ha/y for cultivated areas. The contrary is true for the nitrogen; only in forested areas are nitrate losses as low as 1 kg/ha/y. In general, the losses of nitrate in agricultural areas of Scandinavia, Great Britain and the United States fall between 10 and 25 kg/ha/y.

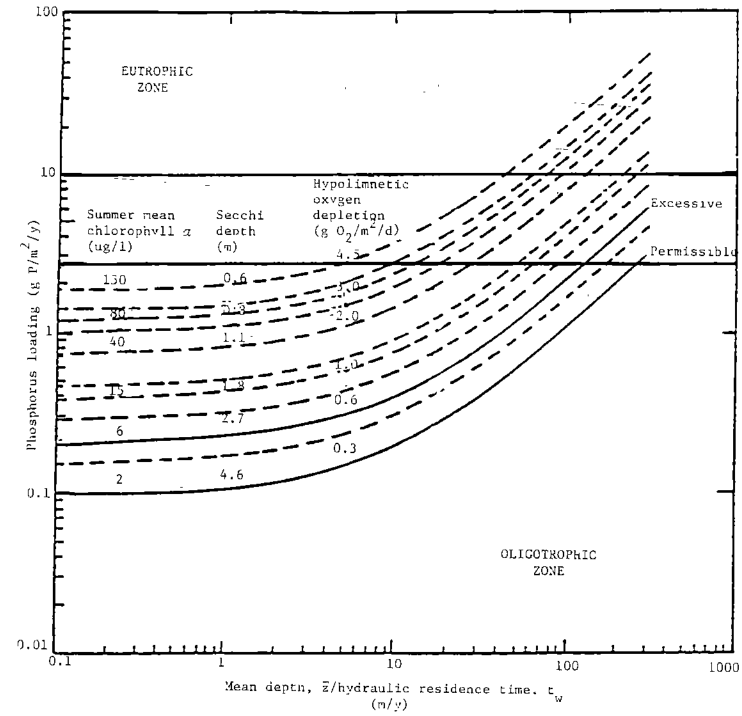
3. LIMNOLOGICAL STUDIES - ABIOTIC FACTORS

3.1 Methodology

The parameters studied were:

- stratification of the temperature, pH, dissolved oxygen,

Fig. 2. *Curves showing the relations between phosphorus load and biological response for a lake* (After [2])



hydrogen sulfide and nitrate with a periodicity of 1 week to 1 month;

- complete chemistry in the surface and bottom waters at three sampling stations once each month;

- horizontal and vertical homogeneity of the water in the reservoir during both the mixed and stratified periods;

- sedimentation rate of detritus and studies of the sediments.

The following methods were used for determining the different parameters:

- water transparency: Secchi disc readings (disk diameter, 30 cm);

- pH: glass electrode;

- alkalinity: TA-titration with fenolphthalein (pH = 8.3); TAC-titration with methyl orange (pH = 5);

- orthophosphate: colorimetric determination after formation by molybdenum blue;

- total phosphorus: mineralization in an acid surrounding followed by colorimetric determination;

- ammonium: colorimetric determination by the blue indophenol method;

- Kjeldahl nitrogen: mineralization using the micro-Kjeldahl method followed by colorimetric determination;

- nitrite: colorimetric determination after the formation of a coloured diazotic component;

- nitrate: determination as for nitrite after reduction by cadmium (after February 1979 the sum of nitrate and nitrite has usually been determined directly);

- dissolved oxygen and BOD_5: Winkler titration (the sample was not aerated prior to incubation for the determination of BOD_5);

- suspended matter: filtration by GF/C glass fibre filters followed by drying at 105 °C and weighing; the organic fraction was determined after heating to 550 °C;

- Oxidability: the permanganate method;

- Sulfur: determination by the methylene blue method after acidification and distillation, or by an electrode specific for S^{2-} ions;

- chlorine: titrimetric determination;

- sulfate: colorimetric determination with methylene blue;

- cations (Na, K, Ca, Mg, Fe, Mn): atomic adsorption spectrophotometry.

3.2 Results and discussion

3.2.1 Water quality studies

To get an idea of the differences between the water quality at the three principal stations, two parameters, one conservative (conductivity) and one non-conservative (dissolved oxygen), were selected for a detailed comparison. No systematic differences between the three sampling stations were found. Therefore, to facilitate reading, only the data from the station near the dam will be presented.

3.2.1.1 Temperature, pH and alkalinity

The temperature of the water varies in general between 12 and 28 °C with only minor seasonal variations between years. During February-March, the water mass is essentially homothermic at a temperature of about 15 °C. In February or March, the water temperature rises and a thermal stratification begins to develop. The temperature of the epilimnion increases about 2 °C/mo during the spring and reaches a maximum in September. The temperature of the hypolimnion raises slowly to about 17 °C. The surface temperature falls after September, and when it approaches the temperature of the hypolimnion by November-December, the reservoir again becomes isothermal and well mixed. Between December and February, the homothermal water mass is cooled to its minimum temperatures (about 12-13 °C).

The principal factors determining the variations in the water temperature are the insolation and evaporation. Inflows have a very limited influence on the temperature. The only evident effects of the inflows on the temperature occurred in connection with the important floods in February 1979 when the minimum temperature of the hypolimnion was attained a couple of weeks later than usual.

The pH also shows variations that are very similar between different years; pH values of 9-10 and 8 occur regularly during the summer and winter months, respectively. Both the pH and the alkalinity depend on the assimilation of CO_2 by the algae. The maximum value of the pH - 9.8 - was registered during a strong algal bloom.

The alkalinity varies inversely with the pH and falls during periods of high algal productivity. The important floods also tend to decrease the alkalinity in the reservoir, while the degradation of organic matter during the autumn tends to increase it.

3.2.1.2 Light penetration

Parameters determining the degree of light penetration are the Secchi disc depth, suspended matter and the turbidity (measured by a turbidometer). The Secchi disc depth varies between 0.2 and 4 m during the period. The highest values occur during weak inflows in the winter season.

The suspended matter shows in general a fairly regular variation (between 0 and 50 mg/l). The organic fraction of the suspended matter varies enormously during the year. During the summer months, when the variation is somewhat less pronounced, the organic fraction has been estimated to constitute 40-80% of the total suspended matter.

Turbidity is a much more reliable quantity than the suspended matter. The correlation between the turbidity and the Secchi disc depth is -0.57 and the regression is described by the following equation:

$$\text{Secchi disc depth (M)} = -0.063 \times \text{JTU} + 2.1$$

No correlation was found between suspended matter and Secchi disc depth [1].

Variations in the particle content in the water are better assessed by measuring the turbidity than by direct analyses. The correlation coefficient obtained for 207 pairs of determinations of the suspended matter and the turbidity was 0.69.

Assessing light transmission in the reservoir, at least for 1979, the relation between the Secchi disc depth and the depth corresponding to 1% of the light incident on the surface could be expressed as:

$$2.56 \times \text{Secchi disc depth} = z(1\%)$$

3.2.1.3 Oxidizing matter

The two principal sources of oxygen in the reservoir are the dissolved oxygen and the nitrate. If both dissolved oxygen and nitrate are lacking, that is, the demand on oxygen by the degrading processes is not met by these two sources, another source, the sulfate, is recruited. This substance is to be avoided, as the consequence of this type of biological process is the formation of hydrogen sulfide.

The dissolved oxygen concentration in the epilimnion usually varied between 14 mg/l and 4 mg/l, but extreme values were encountered: 16.8 mg/l on 19 March 1982 during a very strong bloom of *Peridinium* and less than 0.5 mg/l during the mixing of the water in the reservoir in December 1980. A rapid decrease in the dissolved oxygen concentration in the hypolimnion occurred during the stratified period. It is, however, an important difference that distinguishes 1979 from all other years studied: except for 1979, in all the other years, an anaerobic situation entailing a production of hydrogen sulfide developed at the end of the summer. During that year, traces of oxygen remained in the hypolimnion during the whole stratified period and no hydrogen sulfide was detected.

Quantitative determinations of hydrogen sulfide began in September 1980. Hydrogen sulfide was found during the whole period of April-November 1981. The highest concentrations were, during June-September, generally found at depths between 15 and 20 m, a fact which indicates that the demand on oxidative material was highest in, or right beneath, the metalimnion.

The maximum concentrations of the nitrate fraction of the total nitrogen are determined entirely by the external loads during the winter. The important total inflows during 1977 and 1979 resulted in nitrate concentrations of about 3 mg/l, while the small inflows in 1980 and 1981 yielded only concentrations of 1.2 and 0.5 mg/l, respectively. During the spring and the summer of 1980 and 1981, the nitrate in both the hypolimnion and the epilimnion was completely exhausted, while small nitrate concentrations were found in the reservoir in the autumn of 1979. The details and the consequences of these differences and a more-detailed discussion of the mass balance of the oxidative materials and of the hydrogen sulfide production are presented in section 5.

3.2.1.4 Nutrients

Although nitrates are nutrients, they will be discussed in the context of oxidative materials where they play their principal role. This section focuses on phosphorus and ammonium.

With the exception of the period just after the floods, the total phosphorus concentrations vary between 0 and 100 ug/l with a mean value of about 30 ug/l. The most likely mechanisms behind the fast transport of phosphorus out of the productive zone in the reservoir and its incorporation in the sediments were discussed in section 2.2.3. However, the possibilities of a remobilization from the sediments must also be considered (although phosphate is firmly bound in the sediments under normal conditions). In fact, an increase of the phosphate concentration in the hypolimnion was observed during anaerobic conditions in December 1978, which indicates a certain phosphate mobilization from the sediments at that time. The phosphate concentrations in the epilimnion remain, however, in general quite low during the whole productive period (<15 ug/l).

The maximum concentrations of ammonium during the period were around 0.3 mg/l and 0.7 mg/l in the epilimnion and in the hypolimnion, respectively. A certain increase in the hypolimnic concentrations of ammonium was observed during the autumn of 1978 and 1979, which coincided with the mineralization of the sedimented organic material. During the period of high algal production, the concentrations of ammonium in the epilimnion remained in general below 0.1 mg/l. The results of the analyses of different nitrogen fractions are summarized in Table 3.

Table 3. *Concentrations of ammonium in the epilimnion*

Fraction	Concentrations (mg/l) Mean	Maximum
Inorganic N	0.60	3.4
NO_3	0.50	3.0
NO_2	0.02	0.05
NH_4	0.05	0.3
Organic N	0.50	1.1
Total	1.1	4.5

3.2.1.5 Electrolytes in the water

Conductivity, and thus the total concentration of dissolved salts in the water, show an annual cycle that is largely determined by the inflow. Superimposed on this cycle is a tendency for the conductivity to increase during periods of prolonged droughts. The large flood in 1979 lowered the conductivity in the reservoir to 300 ug/cm. It subsequently increased steadily to reach 900 us/cm at the end of 1980. In particular, after the floods, low values of the conductivity are found in the deep water layers in the reservoir, a fact indicating that the density of the water depends more on suspended solids and the temperature than on the concentration of dissolved inorganic salts. The major factors determining the variations in the conductivity are the origin of the water entering the reservoir, the evaporation and, to a lesser extent, atmospheric deposition of marine salts.

Nearly all the water transported by the rivers to the reservoir during large floods is surface water. Consequently, it is relatively poor in salts dissolved from the soil. During dry periods, a large fraction of the water entering the reservoir is groundwater carrying relatively high salt concentrations.

During long drought periods, a large fraction of the precipitate water infiltrates the soil and dissolves comparatively large quantities of salts, which in turn leads to an increase in the conductivity of the reservoir water. At the same time, the evaporation causes an increase in the salt concentrations near the surface. Evaporation was responsible for about 20% of the increase in the conductivity (about 15 us/cm/mo) during the summer of 1980. This contribution to the conductivity is evidently more important in the epilimnion than in the hypolimnion.

Chlorine and sodium contribute most to the salinity in the reservoir. Although the relation between Na^+ and Cl^- is nearly the same as in the sea, atmospheric deposition probably does not contribute significantly to the salinity of the reservoir in comparison with the contributions from saline groundwater, probably of marine origin, from the Maaziz region.

3.2.2 Homogeneity of the water mass

The most important factor in determining the development of the water quality in the reservoir is probably the stratification. The reservoir is monomictic, with a stratified period that in general extends from the beginning of April until December, when mixing usually occurs.

The thermocline is formed at a depth of about 5-10 meters. During the spring, it descends gradually to a depth of between 10 and 15 meters in June and to a level below 15 meters in September-October.

The thermocline forms a barrier which hinders the transport of oxygenated water from the epilimnion towards the hypolimnion, resulting in an oxygen deficit in the hypolimnion as the degradation of organic material proceeds.

During the summer, a second superimposed stratification cycle often occurs, at a few meters depth and with a frequency of 24 hours. This phenomenon of "atelomixis" is a consequence of the insolation during the day and a wind-induced mixing during the afternoon and night.

3.2.3 Sedimentation rates

The quantity of sedimented organic material largely determines the quantities of dissolved oxygen and nitrate consumed in the hypolimnion during the stratified period. Thus, the sedimentation rate in the reservoir is important information. Sedimentation traps suspended at three depths in the reservoir were used to estimate this parameter during the summer of 1980.

The sedimentation rates of organic matter and chlorophyll were thus estimated at 3-6 $g/m^2/d$ and 5 $mg/m^2/d$, respectively. Given that the total fraction of chlorophyll *a* in the algae is in the range of 0.1 to 0.3%, the sedimentation rate for the algae can be estimated to be 2-5 $g/m^2/d$ (net weight). Assuming the carbon content of the algae to be 20%, the sedimentation rate of algal carbon is 0.4-1 $g/m^2/d$. The primary production of carbon during the same period was 2-3 $g/m^2/d$ (see section 4.1.3) which signifies that 15-30% of the produced cells sinks towards the hypolimnion. The rest, 70-85%, is thus decomposed in the metalimnion and their nutrient content is recycled in the productive zone of the reservoir.

3.2.4 Studies of the sediments

The sediment samples, taken at a depth of 20-30 m during December 1978, show a structure with brown and black layers with thicknesses of about 3 cm. The brown layers probably reflect sedimentation during the floods while the black layers, which are richer in organic material than the brown ones (9-11% vs. 7-8%), probably result from sedimentation during the summer and during the rainy period (the sedimentation of organic matter produced in the reservoir proper probably amounts only to a few millimeters per year and cannot alone give raise to all the material in the black sediment layers). From the sediment studies, the total yearly sedimentation during the first years in the life of the reservoir was of the order of 7 cm/y, which corresponds to about 1-1.5 million tons of sedimented matter. The sedimentation during 1979 was, however, probably higher due to the important floods that year.

The mean phosphate content in the sediment surface was 0.18 mg/g of dry sediment.

4. LIMNOLOGICAL STUDIES - BIOLOGICAL ASPECTS

The biological studies in the Sidi Mohammed Ben Abdellah reservoir covered phytoplankton, zooplankton and fish. Aquatic plants (macrophytes) are quantitatively unimportant in the reservoir, apparently because of the very steep littoral zone and the large fluctuations in the water levels.

4.1 Phytoplankton studies

4.1.1 Species composition

The dominating species in the reservoir in terms of biomass is usually *Peridinium cinctum* although the most abundant species in terms of the number of individuals are the Chlorophyceae - *Cosmarium* spp., *Oocystis* spp. and *Staurastrum* spp. During certain periods the Cyanophyseae (*Microcystis aeruginosa*, *Oscillatoria* sp., *Anabena* spp. and *Aphanizomenon* sp.) may also contribute considerably to the phytoplankton population.

4.1.2 Evolution of the biomass

The cellular volumes of the most abundant species were determined in order to calculate the total biomass from the number of cells in each sample. The following cellular volumes were used in these calculations:

Genus	Cell volume ($mm^3 \times 10^{-6}$)
Ceratium	68
Peridinium	30
Staurastrum	3.6
Cosmarium	2.6
Oocystis	0.6

Using the assumption of unit density for the algae, total algal volumes can be transformed directly to wet weight.

The highest biomass observed was found in April 1978 (110 mg/l). Observed maxima for the last 3 years are 26 mg/l in June 1980, 29 mg/l in June 1981 and 41 mg/l in April 1982. During 1980-1982, *Peridinium* composed 70-85% of the maximum biomass. During the autumn, however, the Cyanophyseae also contributed significantly to the total biomass.

The biomass also was estimated by determining the chlorophyll *a* content (including the phaeopigments) in a water column from the surface to the double Secchi disc depth. The maximum concentrations of chlorophyll *a* were recorded on March 1978 (140 ug/l), June 1979 (220 ug/l) and April 1982

(150 ug/l). Disregarding the extreme values, the maximum chlorophyll concentrations at a depth of 0-5 m varied between 40 and 60 ug/l in 1979, between 20 and 40 ug/l in 1980, around 20 ug/l in 1981 and between 40 and 80 ug/l during the spring of 1982. The mean concentrations in the trophogenic zone varied between 5 and 20 ug/l during the productive period in 1979-1981. Since the depth of the productive zone varies during the year, it may be reasonable to present the chlorophyll *a* content in the water in ug/m^2 rather than a concentration. Figure 3 a-c shows the biomass/m^2 for the period 1980-1982. The vertical chlorophyll distribution in 1982 is shown in Figure 4. If the chlorophyll *a* values are compared with the biomasses estimated from cell counts for a period when *Peridinium* is the dominant algal species (9 April 1982), the chlorophyll *a* content in *Peridinium* is 0.3% of the wet weight. This value corresponds well with that found by Polligher & Berman [4] as typical for Peridineae.

4.1.3 Primary production

The primary production was measured by two methods, assimilation of carbon 14 and oxygen production, from March 1978 to June 1981. After June 1981, only the oxygen production method during 24 hours has been used for the production determinations.

A comparison shows a satisfactory correspondence between the results obtained with the two methods, given a net production of 60-70% of the gross production, a daily production 8-10 times higher than the hourly production at noon and a photosynthetic ratio of 1-1.2. A photosynthetic ratio close to 1 seems reasonable during periods of *Peridinium* dominance since this species is particularly rich in glucides.

Carbon production is generally less than 400 mg/m^2/d during the winter. The production was negligible just after the large flood in February 1979 because of the low transparency of the water at that time, a fact which also retarded the maximum production until mid-summer that year. On the other hand, the primary production increased rapidly already in March in 1981 and 1982, 2 years without any important floods during the winter. At the beginning of May in 1981 and 1982, the production of carbon reached values of almost 5 g/m^2/d and 6 g/m^2/d, respectively.

The vertical distribution of the primary production (measured by the oxygen production method) in 1982 is shown in Figure 5. The production rate displays a higher temporal uniformity than the biomass, a fact which shows that the biomass is

Fig. 3. *Biomass in the reservoir, 1980-1982*

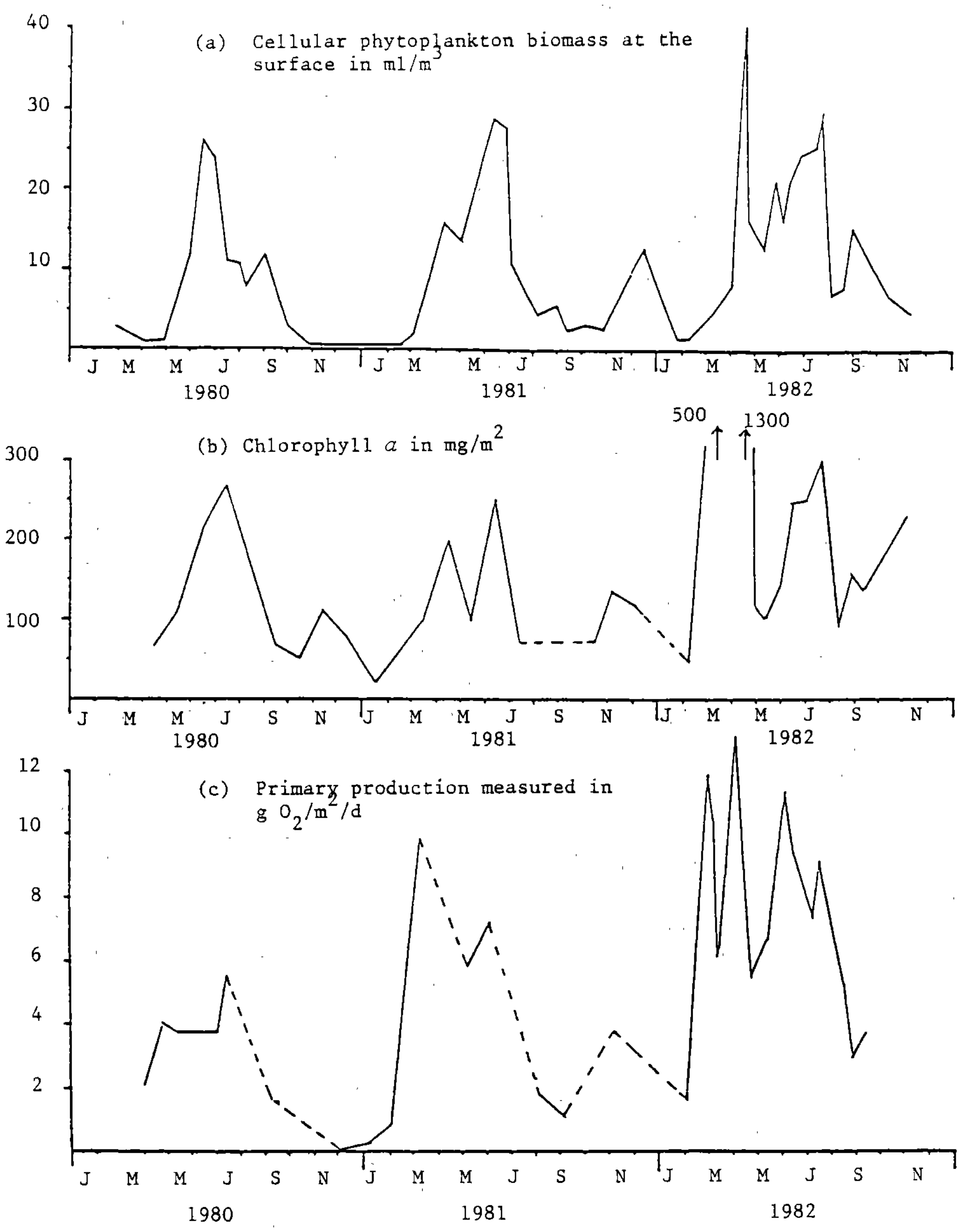


Fig. 4. *Vertical distribution of chlorophyll a during 1982*

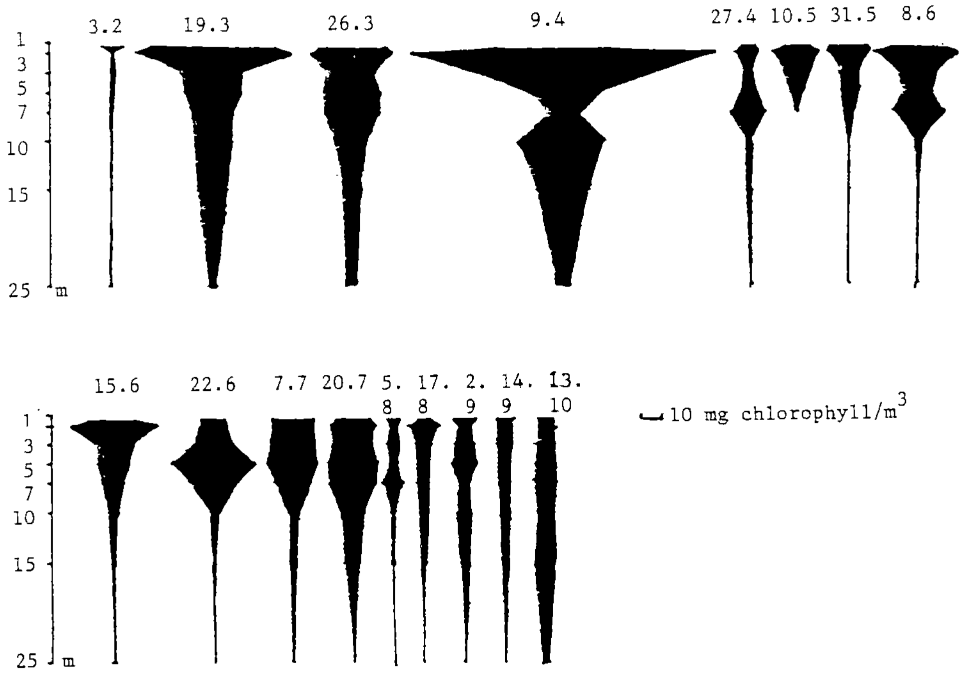
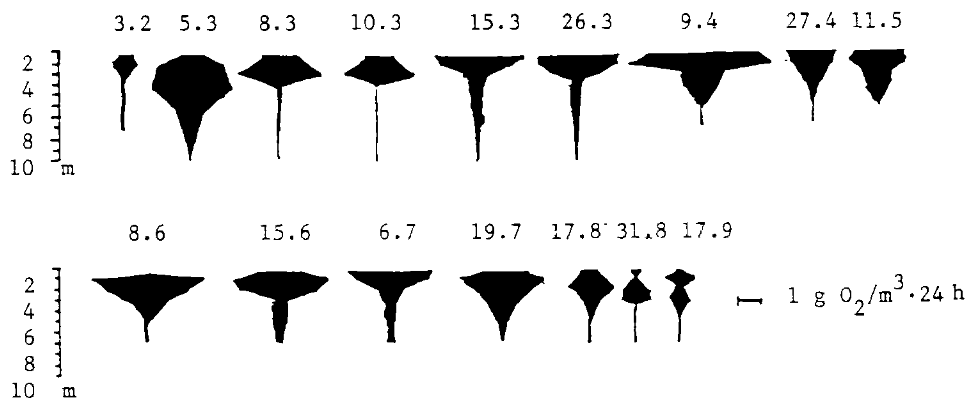


Fig. 5. *Vertical distribution of primary production during 1982*



subjected to controlling factors which are more variable than those determining the production.

Nevertheless, the primary production measurements can be verified during short periods when the biomass increases rapidly by studying the correspondence between the augmentation in the cell volume or the chlorophyll *a* and the directly measured primary production. A period which satisfies the necessary conditions is 26 March to 9 April 1982. During these 14 days, the chlorophyll content and the biomass increased to about 1000 mg/m^2 and 300-400 g/m^2, respectively. Given that the algal biomass consisted almost entirely of *Peridinium* (95%) at that time, the carbon and the chlorophyll content in the algae can be estimated to 20% and 0.3% of the wet weight, respectively. A daily increase of carbon of 4.5 g/m^2 can be deduced from the chlorophyll *a* values. If the biomass results are used instead, the estimated carbon fixation rate is between 4.3 and 5.7 $g/m^2/d$. Net production of oxygen measured at the beginning and the end of the period was 10.4 and 14.1 $g/m^2/d$, respectively, which corresponds to a carbon assimilation of 3.9-5.3 $g/m^2/d$. Thus the direct measurements of the productivity seem to be satisfactory.

The frequency with which the primary production is measured is too low for a reliable calculation of the total annual production. Nevertheless, a very rough estimate of the annual production based on the sparse numbers available can be made. For the years 1979-1982, the values for carbon in $g/m^2/y$ are: 1979, 400; 1980, 300; 1981, 600; and 1982, 700. A high production in 1982 is further verified by several observations of a strong supersaturation of oxygen at the surface (saturations more than 150%). Furthermore, reasonably large inflows occurred late in 1982 (about 180 Mm^3 in April-May), which brought about a direct introduction of nutrients into the epilimnion proper at a time when a fairly important algal population already existed.

4.1.4 Limiting factors for production

The accessible phosphate is often considered to be the most important limiting factor in a lacustrine system. In the Sidi Mohammed Ben Abdellah reservoir, however, light penetration is the principal limiting factor during certain periods (just after important inflows). During most of the productive period, the water has surplus nutrients, in particular as the recycling of phosphorus and nitrogen in the metalimnion is probably quite efficient.

To make a preliminary estimate of which one of the two main nutrients, phosphorus or nitrogen, determines the primary production, one can investigate the N:P ratio in the water during the productive period (Fig. 6). A ratio below 10 indicates that nitrogen and not phosphorus is the limiting factor for the production - unless the capability of direct utilization of atmospheric nitrogen is included as a possibility (certain Cyanophyseae can in fact use the atmospheric nitrogen for growth).

During periods when the N:P ratio falls below 10, Cyanophyseae are found regularly in the reservoir, but during periods when nitrogen is abundant in the water, these nitrogen-fixating organisms are rare. A deficit in inorganic nitrogen (NO_3 and NH_4) thus seems to signify that the dominant algal species, usually *Peridinium* or Chlorophyseae, is at least partly exchanged with Cyanophyseae. During such periods, the total production does not necessarily decrease.

4.2 Zooplankton studies

Relatively few studies of the zooplankton in the reservoir have been carried out. *Diaphanosoma brachyurum* was found to be the dominating species, constituting 80 and 50% of the total population in the first two samples (total biomass 1.1 mg/l and about 0.2 mg/l wet weight, respectively). The remaining identified species were:

Rototoria	*Asplanchna* sp.
	Keratella sp.
Cladocera	*Rosmia longirostis*
	Ceriodaphnia reticulate
	Daphnia cfr. longispina
Copepoda	*Cyclops* spp.

In addition to the zooplankton, *Atyaephyra desmaresti* (Millet), a shrimp, was identified in the reservoir. This crustacean occurred in considerable quantities in the sedimentation traps. At the end of 1981, when the quantitative zooplankton analyses were becoming more regular, the Cladocera (in particular *Diaphanosoma* but sometimes also *Daphnia* and *Bosmina*) predominated over *Cyclops*. Specimens of the rotifers *Keratella* and *Asplancna* also were found occasionally. The total biomass was usually around 0.1 mg/l wet weight. At the beginning of 1982, the larger *Daphnia* were becoming more abundant, and at the same time the total biomass increased. On 6 April 1982, the following composition was found in the water column at a depth between 0 and 6 m:

Fig. 6. *The N/P ratio, biomass and nitrate concentrations as functions of the total inflow during 1976-1981*

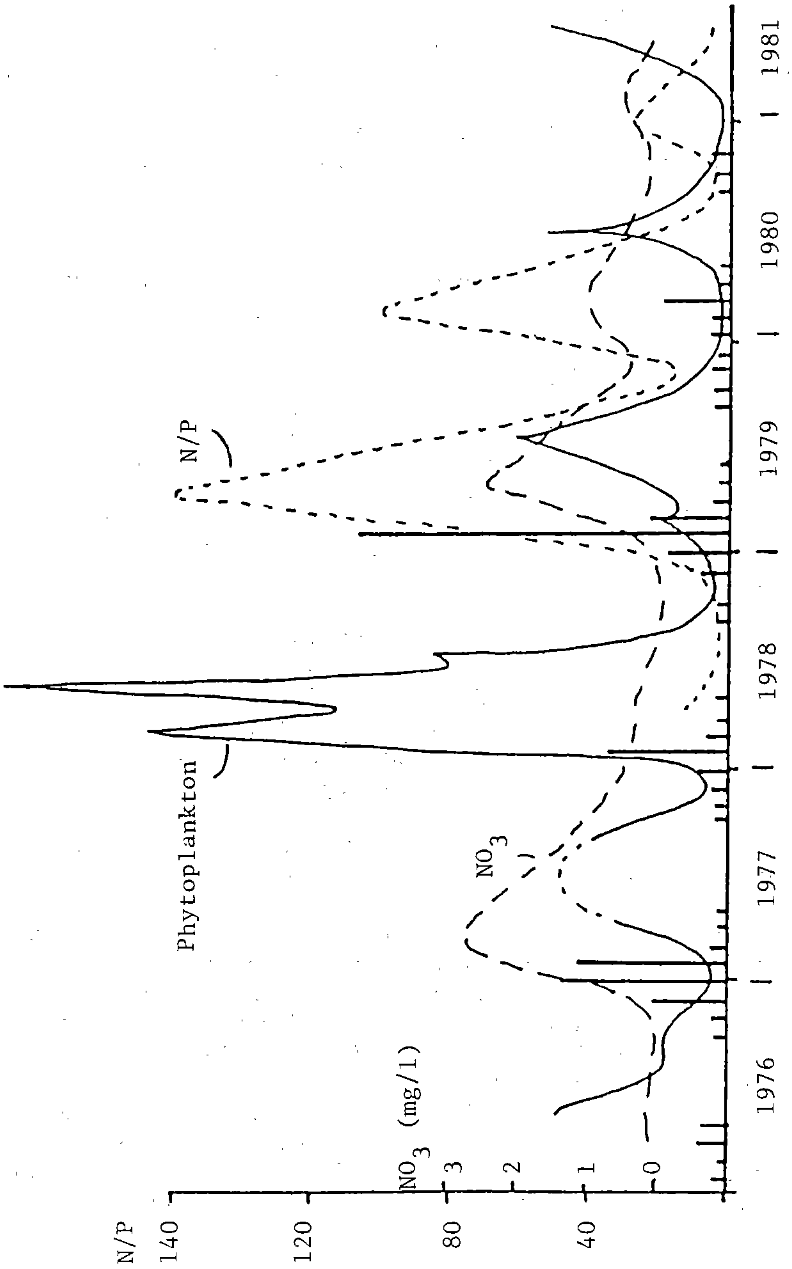


Fig. 6. *The N/P ratio, biomass and nitrate concentrations as functions of the total inflow during 1976-1981*

Group	Biomass (mg/m^3)
Rotifers	10
Ceriodaphnia	180
Daphnia	1700
Diaphanosoma	110
Cyclopoides	900

All the zooplankton groups tripled or more than tripled their abundance between 19 March and 9 April 1982, probably in response to an increase by a factor of three of the algal biomass during the same period. The rapid decrease of the algal biomass that occurred at the end of April was followed by a corresponding decrease in zooplankton biomass (to 85 mg/l). A rapid development of *Cosmarium* followed by a considerable increase of *Daphnia* and *Diaphanosoma* was observed at the end of May, yielding a total zooplankton biomass of 2.5 mg/l on 24 May. The ratio of algal to zooplankton biomass was about 10:1 on this date. For the zooplankton population to become that dense, it is probable that algal species other than *Peridinium* are important since the cladocers generally are poor consumers of *Peridinium*. Serruya [5] has found that the only species of zooplankton that consumes *Peridinium* in Lake Kinneret is *Asplancha*. The strong dominance of *Peridinium* in the algal community in the Sidi Mohammed Ben Abdellah reservoir may thus be explained by a strong zooplankton control of other algal series.

It is not known why *Asplancha*, which should have very good biotic conditions in the reservoir, does not in general constitute a more important fraction of the zooplankton biomass. However, instances do occur when the *Asplancha* population becomes important; in June 1980, this rotifer constituted one third of the total zooplankton population.

4.3 Fish studies

Identified species, as well as their biomass, are shown in Table 4. Further identified but not quantified were the mullet species *Mugil saliens*, *M. capito* and *M. cephalus*. *Barbus fritschii* heavily dominates the fish population in the reservoir. The trial fish samplings neither established the total fish biomass nor the total fish productivity in the reservoir. The data do provide a basis for further studies; in particular, the effect of different measures against the eutrophication on the fish population may be forecast in future studies.

Table 4. *Identified fish species and biomass*

Species	March 1980			June 1980			October 1980			March 1981		
	N	kg	%	N	kg	%	N	kg	%	N	kg	%
Barbus fritschii	1331	76.8	85	>900	79.7	89	2467	86.7	93	2989	131	95
B. callensis	58	5.8	6.5	74	6.5	7.2	52	3.7	4	11	2.2	1.6
Eupomotis gibbosus	2	0.1	0.1	74	2.1	2.3	51	1.3	1.4	11	0.3	0.2
Cyprinus carpio	1	0.4	0.5	3	1.0	1.2	1	0.3	0	3	1.5	1.1
Alosa finta	12	4.0	4.5	1	0.3	0.4	11	1.2	1.3	16	2.2	1.6
Micropterus salmonides	7	0.5	0.5	0			0				0	
Atherina sp	299	2.3	2.6	1	0.01	0	30	0.2	0.2	53	0.4	0.3
Total		89.9	100		89.6	100		93.4	100		138	100

Today a commercial cooperative fishery, organized by the State Fishing Service, operates in the reservoir. The primary targets are eel (*Anquilla anquilla*), mullet (*Mugil saliens*), black bass (*Micropterus salmoides*) and common carp (*Cyprinus carpio*). Mullets of 1.6-3.4 kg are regularly caught but since this species reproduces in the sea and not in the reservoir, it must be introduced from time to time if its harvesting is to continue in the future.

5. BALANCE OF ORGANIC MATTER AND OXYGEN

The consumption of oxygen, dissolved or chemically bound in nitrates, in the oxidation of organic matter (detritus) can be approximately described as follows:

$$O_2 + CH_2O \longrightarrow CO_2 + H_2O$$

$$2\ NO_3 + 3\ CH_2O \longrightarrow N_2 + 3\ CO_2 + 3\ H_2O$$

The latter process is known as denitrification.

The magnitude of these processes can be assessed by studying the diagrams describing the oxygen deficit as a function of depth (see Fig. 7 a-c, which shows the oxygen deficit curves for 1980). The oxygen deficit is expressed as the differences along a profile between the saturation concentration at the actual temperature and the measured concentration. On 23 January 1980, the whole water mass was well mixed and the oxygen deficit was only small. The surface between the full and the pointed curves shows the total amount of oxygen available, while the surface between the full curve and the zero line illustrates the deficit.

During the spring, an oversaturation of oxygen developed at the surface at the same time as the oxygen deficit in the hypolimnion was becoming more and more pronounced. Another phenomenon can be seen in the vicinity of the thermocline at the end of June: an "anomaly" or "lack" of nitrate. This nitrate anomaly is calculated as the difference between the expected nitrate concentration, as deduced from the mean composition of the organic matter in the water, and the measured nitrate concentration [6]. The occurrence of a nitrate anomaly is generally considered to be a good indicator of denitrification.

The nitrate anomaly, which begins at the thermocline, continues to develop towards the bottom of the reservoir. By mid-August a nitrate anomaly is found in the whole hypolimnion - at that time, all the nitrate has been consumed. From this

Fig. 7a. *Oxygen deficit in the reservoir, January–March 1980*

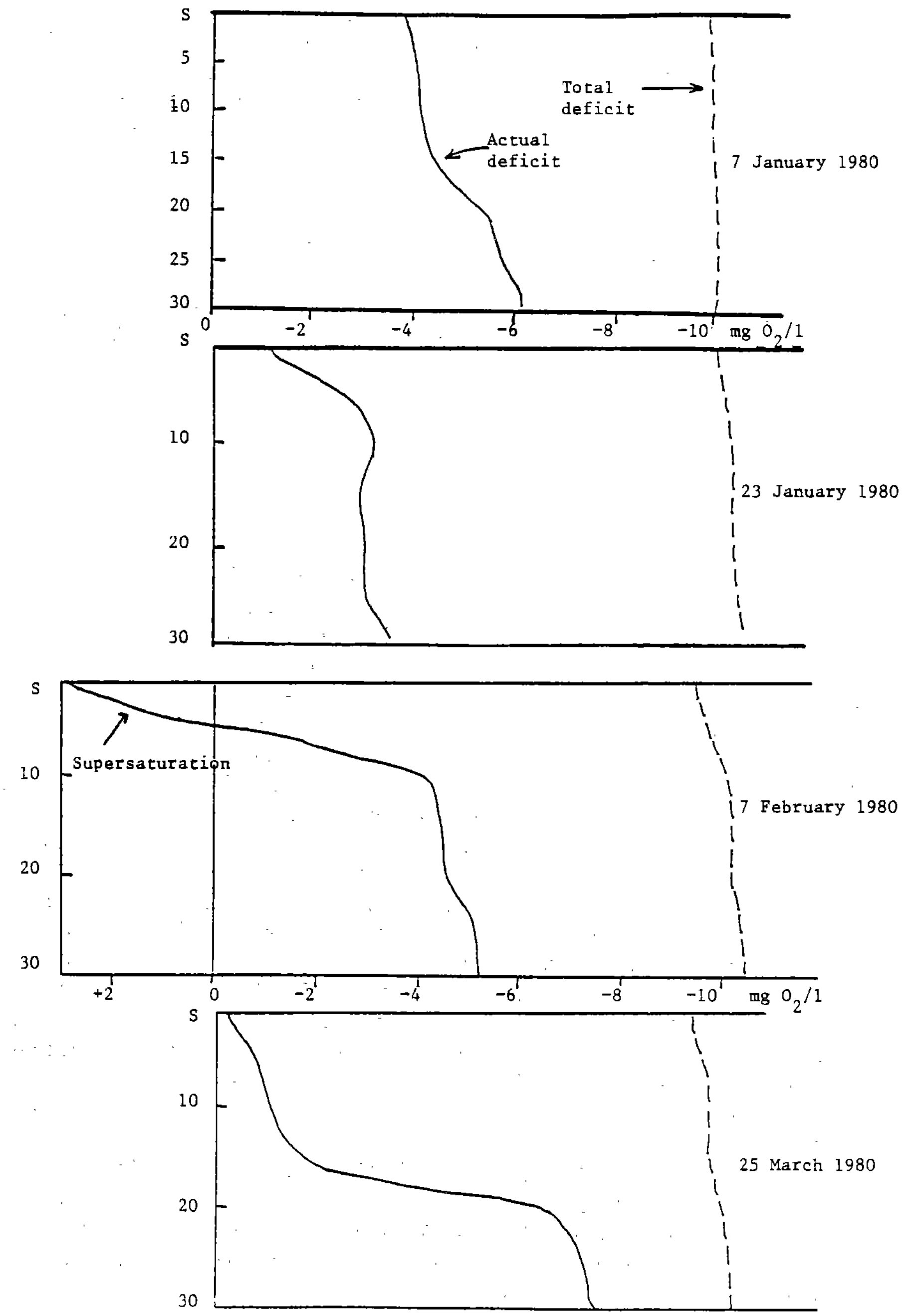


Fig. 7b. *Oxygen deficit in the reservoir, May-July 1980*

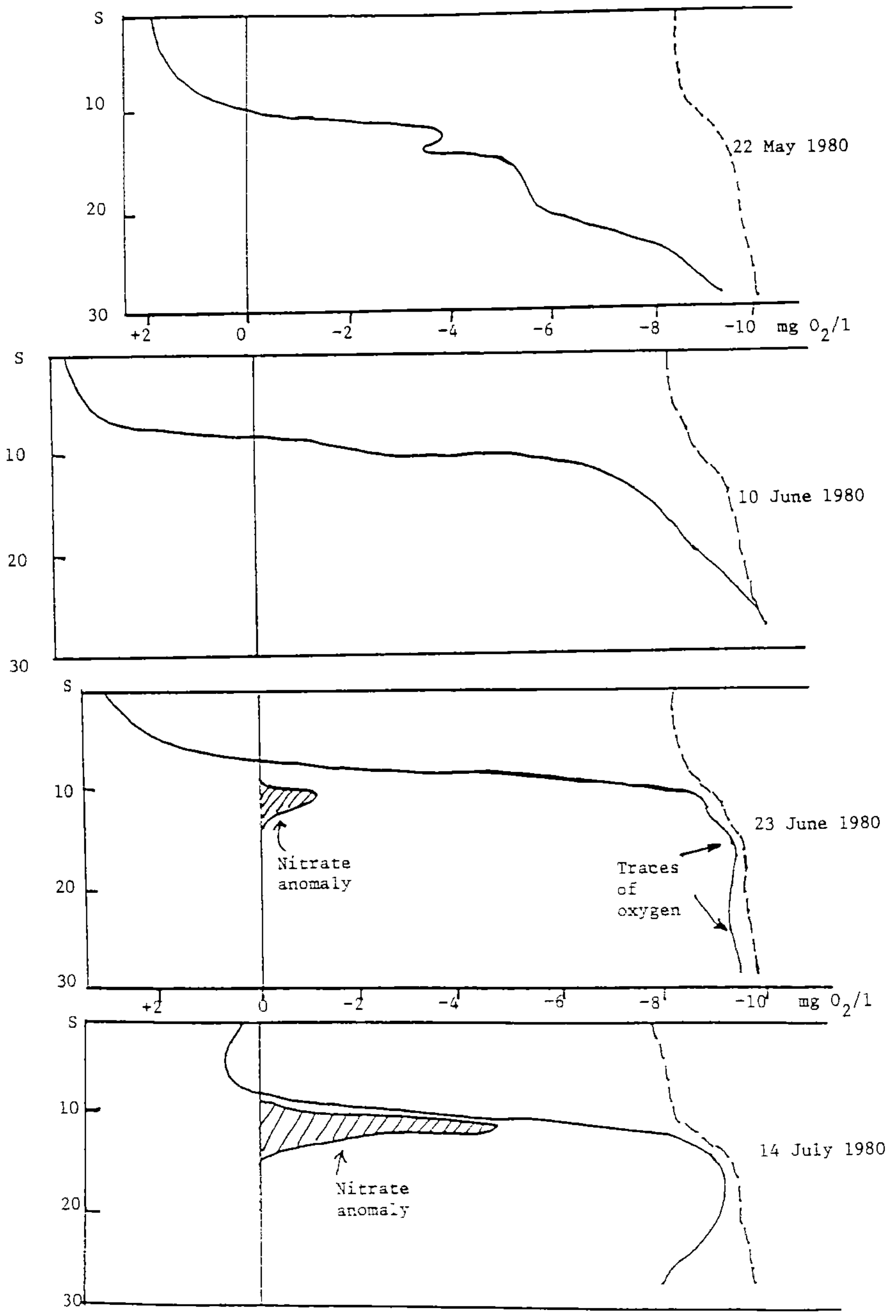
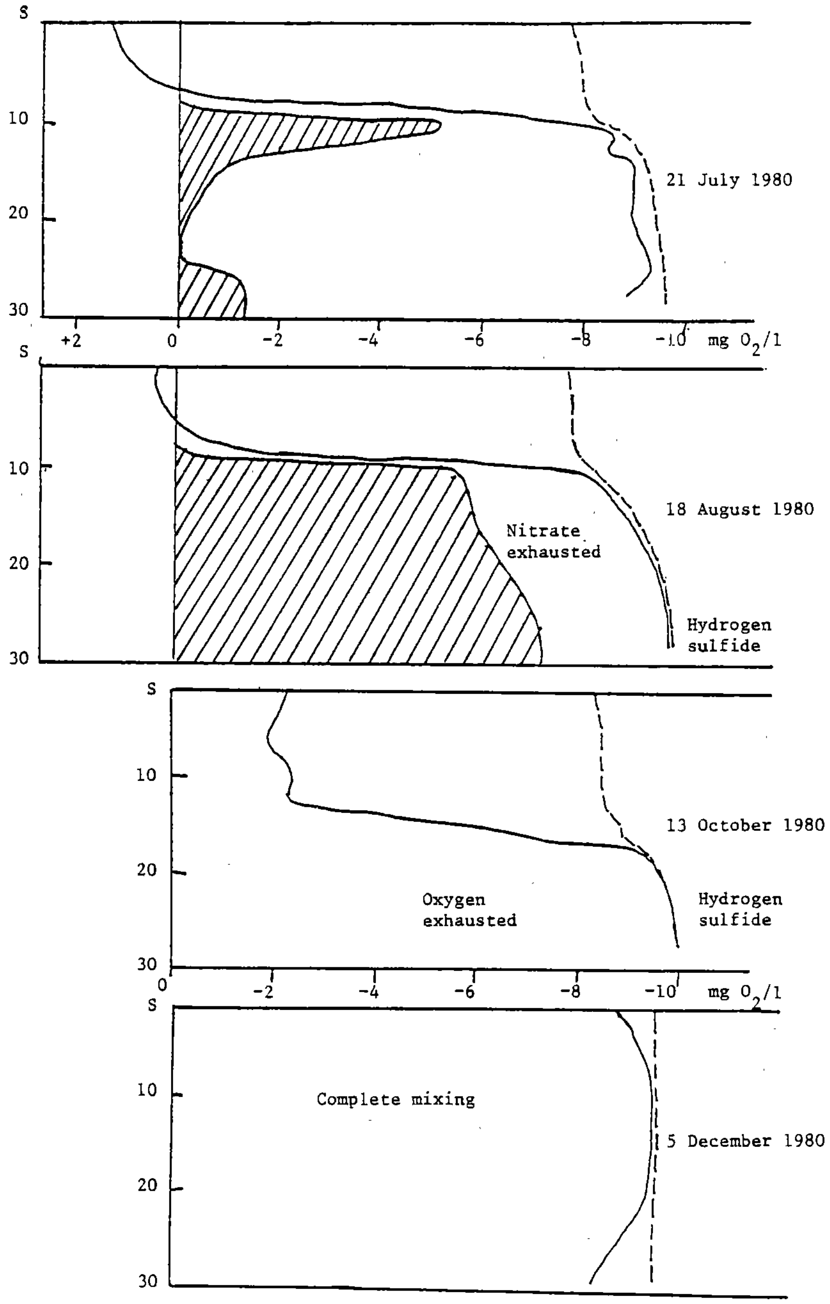


Fig. 7c. *Oxygen deficit in the reservoir, July-December 1980*



moment, the traces of oxygen still remaining in the hypolimnion disappear, a total anaerobicity is established and the production of hydrogen sulfide begins. At the end of the year, the whole water column has an oxygen deficit from the surface to the bottom (see Fig. 7c).

The oxygen is of course consumed in the epilimnion as well as in the hypolimnion. However, the consumed oxygen is continually being replaced by oxygen from the atmosphere and in particular by oxygen produced during the algal primary production. The oxygen consumption is much faster in the epilimnion than in the hypolimnion. Direct measurements in the epilimnion during 15 April to 15 September 1979 yielded an oxygen consumption of about 1.5 mg of dissolved oxygen per day in the first 5 m of the water column. This value corresponds to some 35 000–50 000 tons of oxygen consumed in the whole epilimnion during the period considered. The oxygen-consuming processes in the epilimnion are respiration by algae and zooplankton, and bacterial decomposition of organic material.

In spite of this important oxidation taking place already in the epilimnion, a significant part of the detritus does reach the hypolimnion through the process of sedimentation of suspended organic matter. Since the suspended organic matter plays a decisive role for the water quality in the reservoir, it is important to assess the sedimentation as precisely as possible. The mean amount of particles reaching the hypolimnion during the stratified period in 1979 was measured using sedimentation traps to 3 g/m^2; the carbon value was 0.6 g/m^2, which corresponds to about 4500 tons of particles (wet weight) or 15% of the biomass produced in the epilimnion during the period.

The total exhaustion of oxygen in the hypolimnion is hindered by the presence of nitrates. The nitrates provide a buffer of oxidizing material that retards the use of sulfates as oxidizing agents. The consumption of dissolved oxygen in the hypolimnion was relatively fast (2.4 g^2/mo) during the summer of 1979, and by 15 July only traces of oxygen were left (<0.5 mg/l). At the same time, a nitrogen value of 1.5 mg /l of nitrate was left in the hypolimnion and still in December, traces of nitrate could be found in the water. Consequently, no hydrogen sulfide was observed in the reservoir in 1979.

Unlike 1979, no important floods occurred in the beginning of 1980 and 1981. The situation in 1980 and 1981 was thus entirely different from that in 1979, as illustrated in Table 5.

The consequences of this year-by-year decrease in the amount of dissolved oxygen and nitrate in this hypolimnion at the beginning of the stratified period are very pronounced. As the reserves of oxidizing material in the hypolimnion are consumed, the production of hydrogen sulfide starts earlier and earlier each year. The first traces of hydrogen sulfide were found in August 1980 and already in May 1981, in both cases just after the complete disappearance of nitrate in the hypolimnion.

Table 5. *Conditions in the reservoir in 1980 and 1981*

Year	Inflows during September-March (Mm^3)	Quantity of dissolved oxygen (tons)	Quantity of nitrates (tons)
1979	1540	1300	1400
1980	380	650	800
1981	130	350	400

The situation in 1981 was further aggravated by the fact that the oxygen consumption rate was higher than in 1979, presumably because of a high algal production. The oxygen consumption in the hypolimnion for 1979 was 2.4 $g/m^3/mo$, for 1980 1.5 $g/m^3/mo$ and for 1981 2.9 $g/m^3/mo$. The oxygen consumption in 1981 was thus about 0.095–0.1 $g/m^3/d$, a value which corresponds well with that found in Lake Eire [7]. The variation in the hypolimnic oxygen consumption rate may be compared with the primary production which was (approximately) 400, 300 and 600 $g/m^2/y$ for 1979, 1980 and 1981, respectively.

The consumption rate of nitrate, on the other hand, varied little, remaining around 0.05 $g/m^3/d$ during the whole period.

In 1979, the quantity of oxidizing material originally present in the hypolimnion was sufficient for the mineralization of 900 tons of organic carbon (4500 tons of algal detritus). This corresponds well with the sedimentation estimates for the stratified period that year.

As mentioned above, no hydrogen sulfide was detected during 1979, while hydrogen sulfide was observed in August 1980 in spite of the fact that the primary production was less important in 1980 than in 1979. In 1980, the quantity of oxidizing material originally present in the hypolimnion was sufficient for oxidizing 480 tons of organic carbon. The total amount of sedimented organic carbon was estimated at 600 tons for that year, and sulfates were thus being used in the decomposition processes after the dissolved oxygen and the nitrates had been exhausted in the hypolimnion.

6. MATHEMATICAL MODELS

Eutrophication is a complex process involving physical, chemical and biological factors. A comprehensive analysis of such a complicated system is greatly facilitated by mathematical simulation models. Such models allow, among other things, investigations that are impossible to carry out in the field.

Two types of model can be used in a eutrophication study:

- simple and often empirical models describing the macroscopic compartments in the system using phenomenological equations (often simple and multiple regression polynomials);

- analytical models describing the microscopic compartments in the system using deterministic equations and estimated or measured parameters.

The influence on a water body caused by a change of the load of phosphorous or nitrogen can be estimated by stationary empirical models of the Vollenweider type, but such models cannot determine response times.

The study of the Sidi Mohammed Ben Abdellah reservoir used three different analytical models:

- a one-dimensional hydraulic model essentially based on a standard heat balance formalism (HM-1);

- a two-dimensional hydrodynamic model (HM-2);

- an ecological water quality model (WQM).

The three models treat different types of problem: HM-1 generates the advective currents and the dispersion coefficients during dispersion-dominated periods (periods without important inflows): HM-2 is used for studies during flood periods or for studies of horizontal transport processes. WQM is used for analyses of the temporal development of water quality parameters such as the algal biomass, dissolved oxygen and nitrate. Hydraulic data generated by HM-1 are used as input for WQM.

6.1 Model descriptions

6.1.1 The one-dimensional hydraulic model (HM-1)

The one-dimensional hydraulic model is based on a heat balance formalism supplemented with a simplified mass-balance type description of the advective currents.

The water mass is divided into horizontal elements approximately 1 m deep. The following equation is used for each horizontal element:

$$dc/dt \times dV = d_z \times \bar{Q}_z \times dc/d_z + d_z \times A_z \times D_z \times d^2c/d_z^2 + Q_{in} \times C_{in} - Q_{out} \times C_{out} + S$$

where c = the heat concentration, dV = the volume of the element (m^3) d_z = the depth (m), Q_z = the vertical advection (m^3/s) A_z = the surface of the element (m^2), D_z = the dispersion coefficients (m^2/s), Q_{in} = the inflow (m^3/s), C_{in} = the heat concentration in the inflow, Q_{out} = the outflow (m^3/s) and S = other sources and sinks for the heat (insolation, reflection, evaporation).

The heat balance is calculated from the heat absorption and emission at the water surface assuming the reservoir to be completely isolated at the bottom. The heat exchange is thus determined by the insolation, reflection at the surface, back-radiation, heat difference between the water at the surface and the atmosphere, and the evaporation. The last factor mentioned is very important for the total heat balance.

On the given scale, the dispersion in the water mass depends on the molecular dispersion, macroscopic turbulences generated by the wind, and on density currents. Mathematically, the dispersion enters the model through a second derivative of the heat concentration. The dispersion coefficients, which appear as factors multiplying the second derivative, are the key factors in the model. These or similar, somewhat descriptive, coefficients are used in all heat balance models.

Two versions of the heat balance model, in which the dispersion coefficients are calculated differently, have been developed for the studies of the Sidi Mohammed Ben Abdellah reservoir. In the first version, the dispersion coefficients are simple functions of the depth and the stability of the water mass. In this case, the wind effect is introduced implicitly through two parameters calibrated against measured temperature profiles. The second version uses a formalism with an explicit dependence of the dispersion coefficients on the horizontal currents and thus on the wind. A dynamic element is thus introduced in the second model version, making the model less dependent on empirical factors, and thus more generalizable but more dependent on high-quality meteorological data.

The advective currents, Q_z in the formula above, are in the model determined by the mass balance in the reservoir and a simple formula describing the distribution of the currents generated by the in- and outflows (this current distribution also can be supplemented in the input to the programme). This very simplified treatment of the advection gives satisfactory results during dispersion-dominated periods but not under winter situations.

The absorption and the reflection of the incident radiation, the back-radiation and the heat exchange with the atmosphere are calculated with standard formulae. The evaporation can be calculated by a standard formula or be supplied in the input or be determined by the mass balance in the reservoir. Other important factors, such as the cloud cover, dry and wet temperatures, winds and the in- and outflows, are provided to the model as external functions.

The integration step used in the model is 3 or at most 6 hours. The reason for using such a short time step (normally a time step of 1 day is sufficient) is the large temperature difference between night and day in Morocco. The cooling of the surface during the nights causes fairly deep regions (4-5 m) of *complete mixing* and the depth of the mixed water layer close to the surface differs greatly between night and day. It is very important to account for these differences when the dispersion coefficients calculated by the HM-1 are to be used by the water quality model.

The number of elements used in the model is fixed, and the depth of each element thus varies with the depth of the reservoir. In the present case, 47 segments were used, and the depth of each segment was thus always in the vicinity of 1 m.

The numerical integration of the equations is carried out with an implicit integration technique.

Thus, the principal quantities calculated by the programme are temperature profiles, dispersion coefficients, element thicknesses and advective currents.

6.1.2 The two-dimensional hydrodynamic model (HM-2)

The two-dimensional hydrodynamic model was developed from an estuary programme originally written by W. Wilmot, IVL, Sweden. The original model has been applied on the Baltic Sea, in the strait between Sweden and Denmark and an estuary in the Baltic Sea. The model was implemented for the case of the Sidi

Mohammed Ben Abdellah reservoir principally to study the effects of an artificial circulation of the water in the reservoir.

The principal equations in the model describe the velocity and density fields in two spatial dimensions and as functions of time. The equations contain approximations which are responsible for simulations of lakes and reservoirs.

The turbulence in a water body can be described, in principle, by an infinite series of coupled equations, which under certain conditions can be simplified to a few equations of low order. The system can be further simplified by replacing these remaining equations entirely by an approximative expression which essentially consists of a product of an eddy diffusion coefficient and a mean gradient. This kind of simplified description is used in the present model.

The vertical water movements are described in the model by equilibrium hydrostatics (mass balance) and by a vertical gradient that is adjusted to the density field. The density is considered constant when multiplied by acceleration or friction terms (the Boussinesq approximation). The water mass is treated as noncompressible.

The change in the depth of the reservoir is not calculated but introduced in the input stream.

An implicit integration technique is used by the model. The length of the time step is calculated, using a stability criterion based on the Friedrich-Courant-Levy number.

The two-dimensional hydrodynamic model is difficult to use and demands substantial computer resources. It has never been implemented on the local computers in Morocco.

6.1.3 The water quality model (WQM)

The water quality model is of a conventional type, composed of a number of coupled non-linear first-order differential equations which are solved by an explicit numerical integration technique. The model is strictly deterministic and thus does not produce any probabilistic information.

A reservoir can be divided in several (horizontal and vertical) segments in the model. The advective currents and the dispersion describing the water exchanges between the segments must be provided as external functions. Only a vertical division of the reservoir has been used in the present study.

The content of phosphorus and nitrogen in algae varies strongly between species and even within a species under different external conditions. The state variable describing the algae has consequently been divided into three compartments (treated in the model as separate state variables) for carbon, nitrogen and phosphorus, respectively. Given that the algae interact strongly with detritus and sediments, the same division into three compartments have been used for the latter state variables. The biomasses of zooplankton and fish are much less important than the algal biomass; furthermore, the time-dependency is less pronounced for the zooplankton and, in particular, for the fish population than for the algae. It has thus not been considered necessary to subdivide zooplankton and fish into separate compartments for carbon, nitrogen and phosphorus; this simplification makes the calculated nutrient budgets approximative. The error introduced into the calculated budget is, however, of minor importance.

The state variables used in the water quality programme are as follows:

- phytoplankton carbon;
- phytoplankton nitrogen;
- phytoplankton phosphorus;
- detritus carbon;
- detritus nitrogen;
- detritus phosphorus;
- sediment carbon;
- sediment nitrogen;
- sediment phosphorus;
- zooplankton;
- fish (algivores);
- ammonium;
- nitrate;
- phosphate;
- dissolved oxygen (hydrogen sulfide).

Hydrogen sulfide is described by negative oxygen concentrations. The model also contains a description of the important denitrification process.

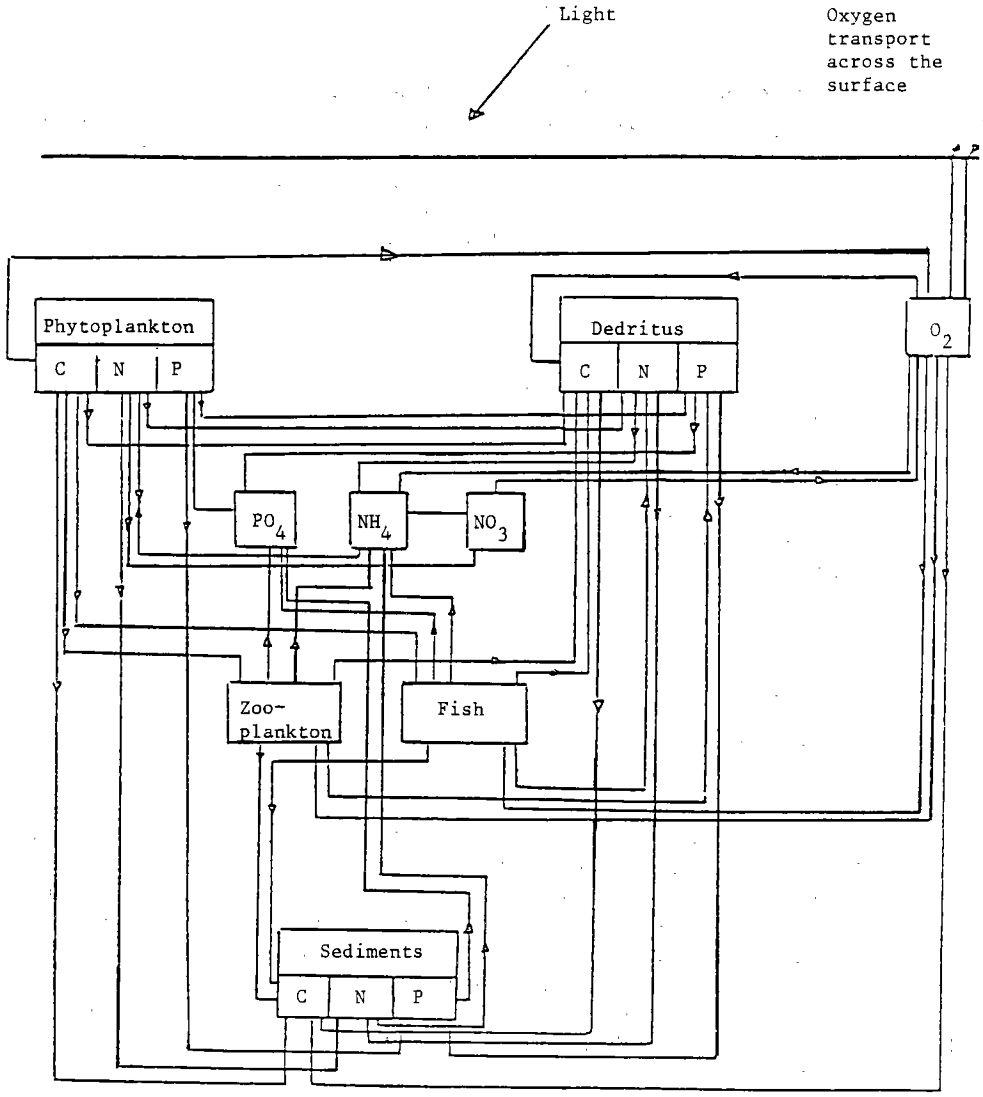
The interactions within a segment are described by different processes in the model. The primary production, the consumption of nutrients by the algae and the consumption of algae by the zooplankton are examples of processes.

The external functions entering the model are insolation, water temperature in each segment, ratio between the surface and the bottom of each segment, volume of each segment, advective currents, exchange volumes (the dispersion), outflows and external loadings.

The external functions are provided to the programme in the form of time series.

The model is schematically described in Figure 8.

Fig. 8. *Principal structure of the water quality model*



The specific functional form of the various processes is based largely on laboratory experiments and experience gained in modelling natural systems. However, the many interactions are of first order when the concentrations are small and of zero order when the concentrations are large. Consequently, most interactions entering the processes are described by Menten-type functions or functions with similar characteristics. The Menten equation has the form $x/(x+a)$ where x is a variable and "a" a parameter (half saturation constant). For $x \ll a$, this equation becomes linear in x while for $x \gg a$, it becomes constant (=1).

In the application of this model to the Sidi Mohammed Ben Abdellah reservoir, the water mass was divided into five segments. With 15 state variables for each segment, a total of 75 state variables was obtained for this case. An explicit integration method, Runge-Kutta or Predictor-Corrector (the choice between the methods is under input control) is used for the numerical integration.

The present model is based on a water quality model developed at the Water Quality Institute (VKI) in Denmark. The most important differences between the original and the present model are the introduction of hydrogen sulfide, modifications in the nutrient uptake processes and of several processes describing grazing and denitrification, and the introduction of a new integration technique (the integration was made in two steps in the original model but this was not possible in the case of the Sidi Mohammed Ben Abdellah reservoir).

Although the ecological model is based on the model developed in Denmark, a completely new programme had to be written in order to implement the model on the local computer in Morocco.

6.1.4 Interface of the one-dimensional hydraulic programme on the water quality programme

While a thickness of about 1 m is used for each horizontal element in the HM-1 programme, a much coarser spatial division is used for WQM. This change in spatial structure does not cause any problems either for the advection or for the temperature or the inflows/outflows calculated by HM-1; the problem is simply to find the desired data in the lists generated by the hydraulic programme and to generate appropriate mean values for the temperature. On the other hand, the dispersion coefficients are associated with a linearized second derivative and depend explicitly in the spatial dimensions. Thus, the dispersion coefficients had to be interpreted by a separate computer program before they could be used by the water quality programme.

6.2 Calibration and testing of the models

6.2.1 The one-dimensional hydraulic model

The model was tested during April-December 1979. Two fairly important floods occurred during this period, 13-17 September and 13-18 October, which together supplied the reservoir with some 75 Mm^3 of water. During an opening of the bottom gate at the end of October, some 50 Mm^3 of water was let out of the reservoir.

The correspondence between the simulated and the measured temperature profiles is good (Fig. 9). It is particularly gratifying that the gradual disappearance of the thermocline during the autumn is well reproduced by the model. The largest deviations between the calculated and measured results occurred on 29 October (Fig. 9). The reason for the comparatively large discrepancy on these data is probably the opening of the bottom gate in October. The currents generated by the water flow through the bottom gate are too strong to be well described by the simple advection formulae used in the model.

6.2.2 The two-dimensional hydrodynamic model

The model HM-2 was used for simulations for 1979. The simulated (two-dimensional) temperature profiles are in satisfactory agreement with measurements.

6.2.3 The water quality model

Figures 10 and 11 show some of the results for 1980.

The primary production seems to be somewhat underestimated by the model. However, the results are difficult to evaluate because of the scarce observations and because of suspected uncertainties in the measurements (only results obtained with the carbon-14 method were used in the comparisons). The aim of the modelling effort is, however, to describe the reservoir on a qualitative or semi-quantitative level and, for that purpose, the results are satisfactory.

The development of the algal biomass is well reproduced in the calculations. The biomass is a simpler variable to model since it does not depend as strongly as the primary production on variations in the external factors and in other variables. The nitrate concentrations in the surface segment are well reproduced. The phosphate concentrations are always very low, which means that the measurements are not entirely reliable.

Fig. 9. *Measured and simulated temperature profiles during 1979*

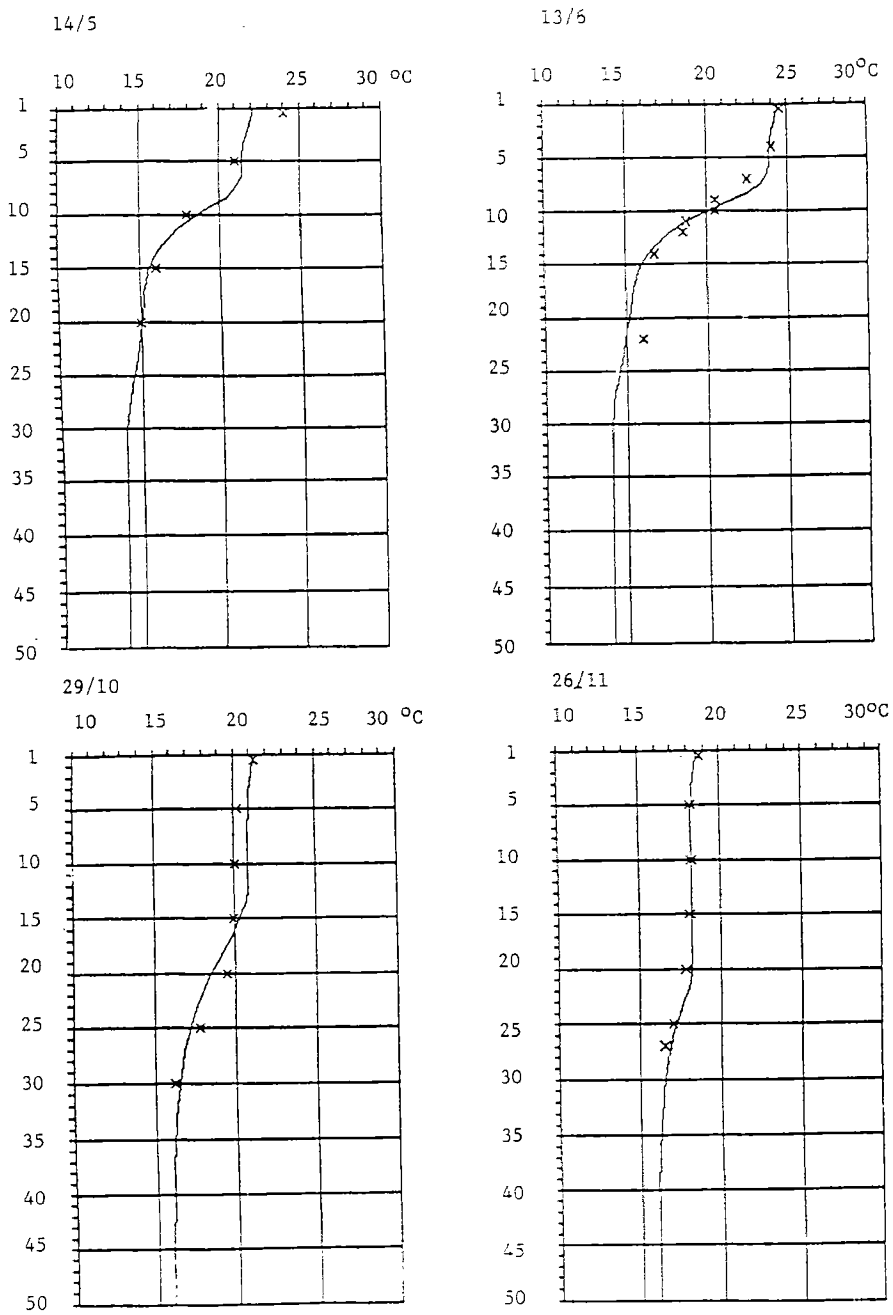
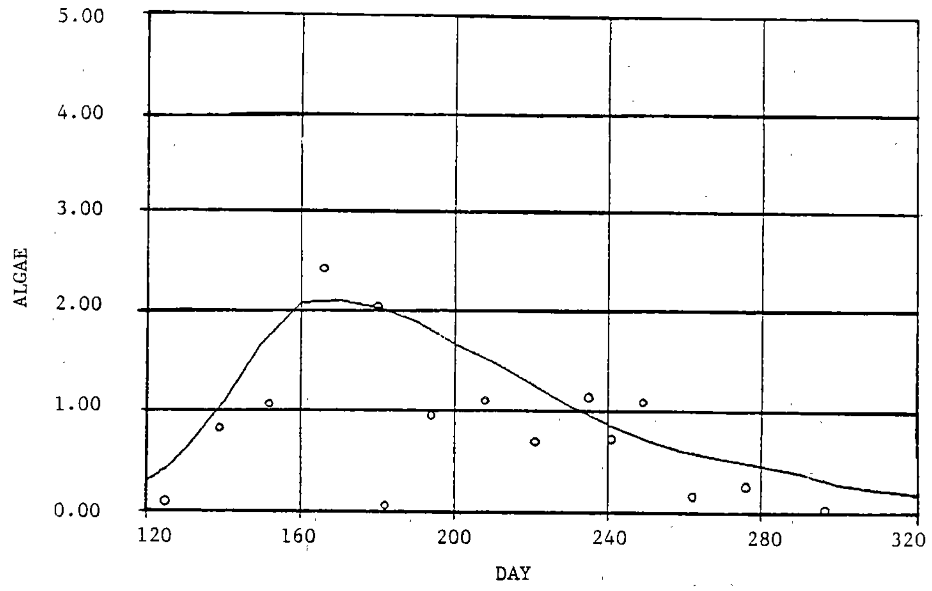


Fig. 10. *Measured and simulated phytoplankton and phosphorus concentrations at the surface for 1980*



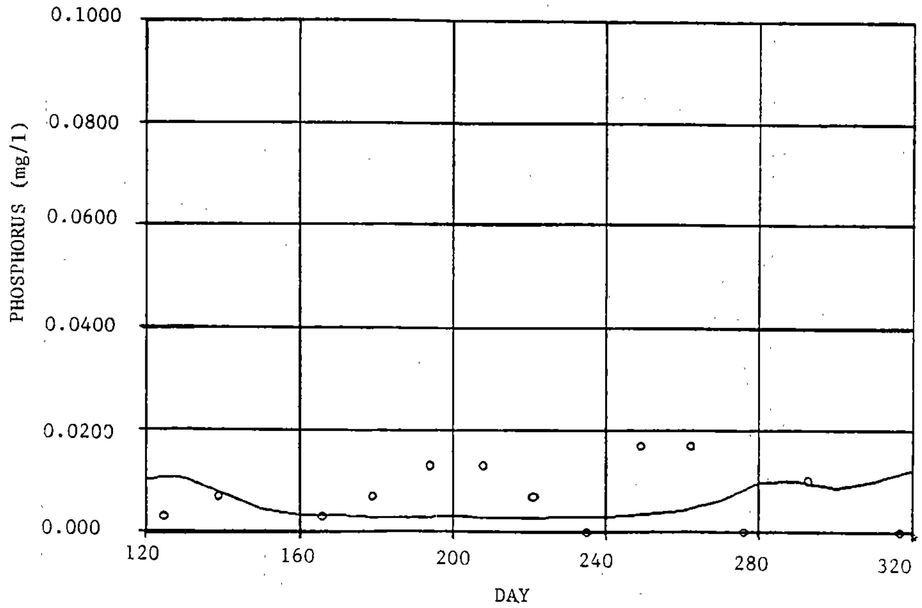
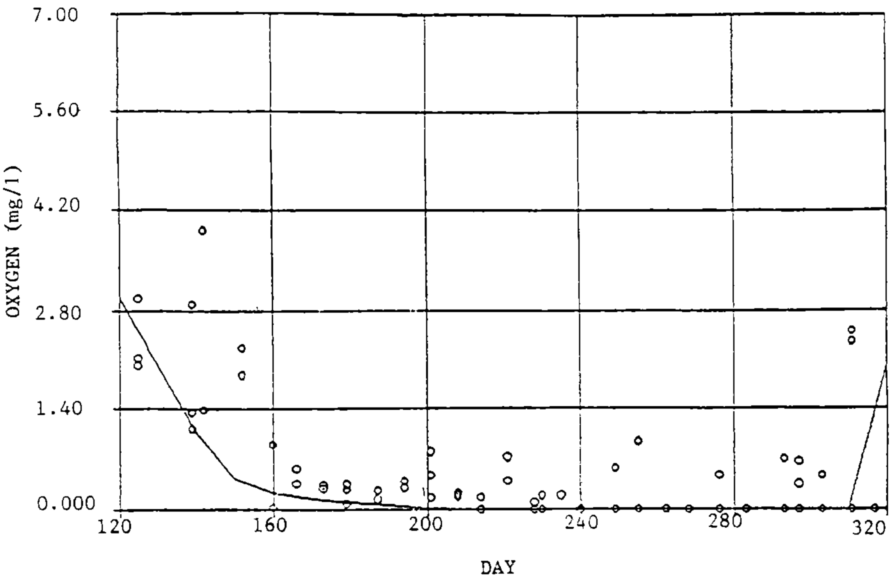
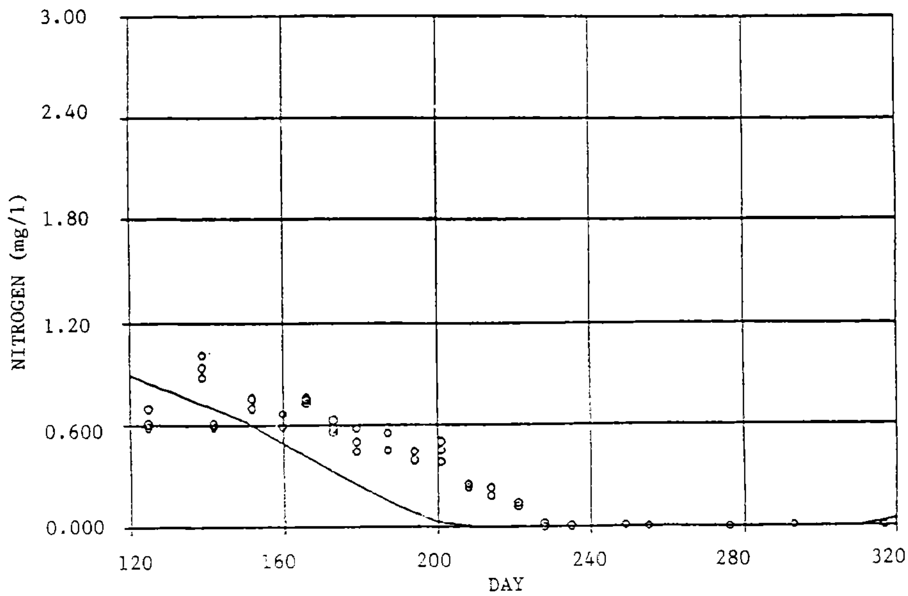


Fig. 11. *Measured and simulated oxygen and nitrate concentrations at the bottom for 1980*





Taking this into consideration, the results obtained for the phosphate concentrations in the surface segment also are satisfactory.

The hypolimnion is largely described by the fifth segment in the model. The correspondence between measured and calculated values for dissolved oxygen and nitrate is good in the hypolimnion. The simulations seem, however, to overestimate the phosphate concentrations, and this discrepancy could not be corrected without using unreasonable parameter values. On the other hand, it is somewhat surprising that the phosphate concentrations remain as low as they do in the hypolimnion, considering the sedimentation rate in the reservoir. It is thus probable that a precipitation of phosphate, presumably in the form of hydroxyapatite, takes place in the hypolimnion. The question of phosphate precipitation in the reservoir also has been discussed earlier in the report.

The results of the simulations are also satisfactory for segments 3 and 4. Segment 2 is not well described because of the strong interaction between segments 1 and 2. Segment 2 is fairly small, and segments 1 and 2 frequently become mixed during the nights. This fact makes it somewhat unreasonable to try to obtain results in good agreement with measurements for the second segment.

Simulations were also performed in 1980. The quality of the results is about the same for 1980 as for 1979. A certain irregularity in the measured values for oxygen and nitrate is found at the beginning of May. This irregularity is probably caused by an opening of the bottom gate at the end of April and/or the occurrence of a short rain period at the same time. The advective currents and the dispersion coefficients from 1979 were used in the simulations for 1980 and the particular conditions at the beginning of May were thus not accounted for in the simulations. It is, however, satisfying that the hydraulic conditions are so similar for the 2 years (except, of course, for the particular short spring period).

REFERENCES

1. Gregory, K.I. & Walling, D.E. Drainage basin form and process. London, Edward Arnold, 1979.

2. Vollenweider, R.A. Advances in defining critical loading levels for phophorus in lake eutrophication. Memorie Istituto Italiano di Idrobiologia Dott Marco de Marchi, 33: 53-83 (1976).

3. Pollingher, U. & Berman, T. Temporal and spatial patterns of dinoflagellate blooms in Lake Kinneret, Israel (1969-1974). Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19: 1370-1382 (1975).

4. Serruya, C. Nitrogen and phosphorus balances and load-biomass relationship in Lake Kinneret (Israel). Internationale Vereinigung für Theoretische und Angewandte Limnologie, 19 1357-1369 (1975).

5. Rönner, U. Denitrifikation i marin miljö In: Processer i kvävets kretslopp. [Denitrification in the marine environment. In: Processes in the nitrogen cycle]. Stockholm, National Swedish Environment Protection Board, 1979.

6. Lee, G.F. et al. Eutrophication of water bodies: Insights for an age-old problem. Environmental science and technology, 12: 900-908 (1978).

SIDI SALEM RESERVOIR

This summary of the evolution of the trophic state of the Sidi Salem reservoir in Tunisia is largely based on a report by Krause [1], established within the Technical Cooperation Programme between Tunisia and Federal Republic of Germany, and on a report prepared by M. Saadaoui within the regional UNDP project, subproject *Eutrophication Control in Lakes and Reservoirs in Northern Africa* (UNDP/RAB/80/011).

1. INTRODUCTION

The Sidi Salem reservoir on the Medjerdah River is situated close to the township of Testour about 60 km southwest of Tunis (Fig. 12). It represents some 50% of the available surface water resources in Tunisia and has fundamental importance to the water supply of the northern parts of the country. Sidi Salem serves as a domestic water resource for Tunis and Cap Bon, and for irrigation in the agricultural northeastern region (Testour, Medjez, Basse Valley, Mornag and Cap Bon).

The Sidi Salem reservoir is located in a fairly populated agricultural region. The pollution sources in the drainage basin of the reservoir thus include diffuse sources from agricultural activities as well as point sources of both domestic and industrial origins.

The most important population centres in the drainage basin, with a population of 10 000 to 50 000 inhabitants, are Béja, Bou Salem, Jendouba, and Le Kef in Tunisia and two more towns in Algeria. In addition, the basin contains a large number of smaller townships with a population of between 5000 and 10 000 inhabitants.

One of the most important characteristics of the climate as regards water quality in the reservoir is the strong variation in precipitation. The mean precipitation in the Béja region during the period 1901-1960 was 645 mm/y, most of which arrived at the end of the dry season. This situation results in a strong erosion as the region has no protecting vegetation cover.

Important data on the reservoir are presented in Table 6.

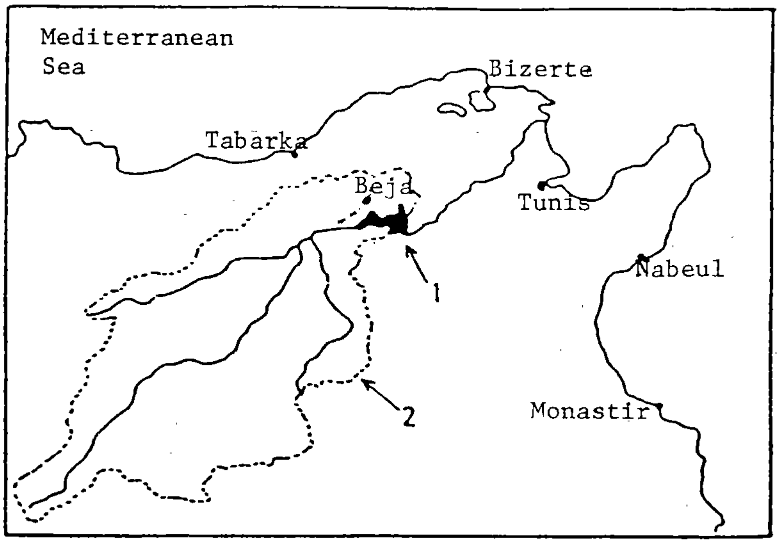
The reservoir has a rather complicated morphology and may be regarded as consisting of three zones (Fig. 13). *Zone 1* may be regarded as part of the Medjerdah River. The depth varies between a few metres to close to 20 m at point 4. The volume of this part of the reservoir is 6.5 Mm^3 at the level of 105 m AMSL, the surface is 975 hectares and the mean depth is 6.7 m. This part of the reservoir is usually well mixed due to

Table 6. *Data on the Sidi Salem reservoir*

Parameter	Value
Surface of drainage basin:	18 250 km^2
Mean inflow	720 Mm^3/y
Maximum inflow	2 000 Mm^3/y
Capacity at normal level:	
until 1985 (AMSL = 100 m)	235 Mm^3
1985-2000 (AMSL = 110 m)	550 Mm^3
Surface at 100 m AMSL	2 330 ha
Surface at 110 m AMSL	4 300 ha
Length of reservoir	42 km
Mean depth at 100 m AMSL	10 m
Mean depth at 110 m AMSL	13 m
Water withdrawal levels:	
Electrical power station	97.5 m AMSL
Irrigation	75 m AMSL
Bottom gate	72.5 m AMSL
Water demands:	
Irrigation	350 Mm^3/y
Domestic water (actual demand)	48 Mm^3/y
Domestic water (future demand)	150 Mm^3/y

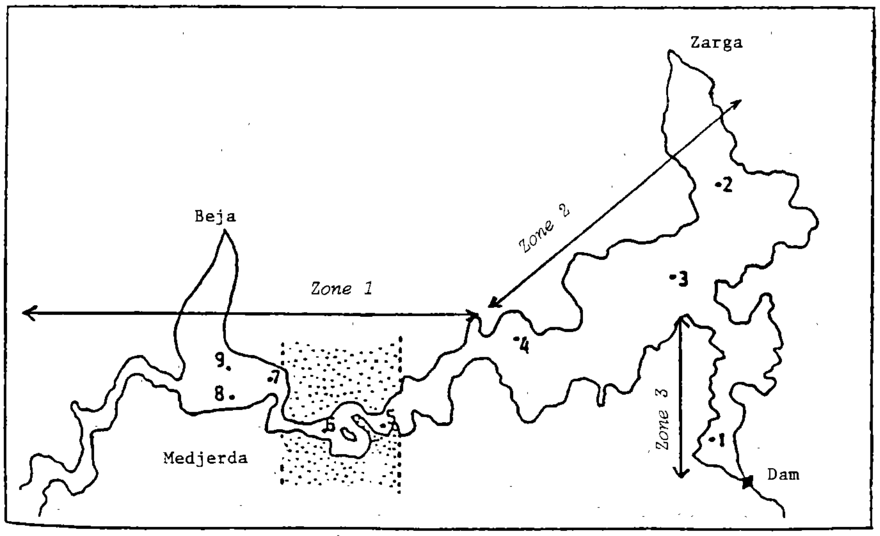
the influence of Medjerdah. Zone 1 has two important nutrient sources: the Medjerdah River and the Béja River (which is strongly polluted by effluents from the town of the same name). *Zone 2* stretches from point 4 to the entry point of the Zarga River. The length of zone 2 is approximately 9 km, including the 3-km-long narrow entry zone for the Zarga River. The volume of this sub-reservoir is approximately 220 Mm^3. *Zone 3* consists of the southern arm of the reservoir and is limited by the dam. At 105 m AMSL, the mean depth is 35 m and the volume is 27.5 Mm^3.

Fig. 12. *Location of the Sidi Salem reservoir*



1 = the dam; 2 = limits of drainage basin

Fig. 13. *Morphology of the Sidi Salem reservoir*



Dotted area indicates saline triassic works
Points 1-9 indicate sampling stations

2. HYDRAULIC BALANCE AND THERMAL STRATIFICATION

The reservoir reached full capacity at the end of 1981.

Sidi Salem is a monomictic and warm reservoir with two periods. The well-mixed *winter period* stretches from November to March. During this period, the water mass is well oxygenated from the surface to the bottom. The stagnant *summer period* stretches from April to October. During this period, the water mass is stratified with an epilimnion 8-10 m deep. The temperature in the epilimnion attains values of 27-30 °C while the temperature of the hypolimnion remains at 12-15 °C during most of the stagnant period. The evolution of the surface and bottom temperature is shown in Figure 14.

3. WATER TRANSPARENCY, DISSOLVED OXYGEN CONCENTRATIONS AND NUTRIENTS

The water transparency has been regularly investigated by means of Secchi disc measurements. The values obtained were less than 2 m during the whole period of 1982-1985. At sampling stations 1-4, the mean transparency was 1.3-1.6 m with occasional maxima between 2.1 and 4.3 m and minima around 0.7 m. The low transparency is not caused by phytoplankton but rather by mineral particles. At the same sampling stations, the concentrations of suspended matter varied between 1 and 28 mg/1 during the same period.

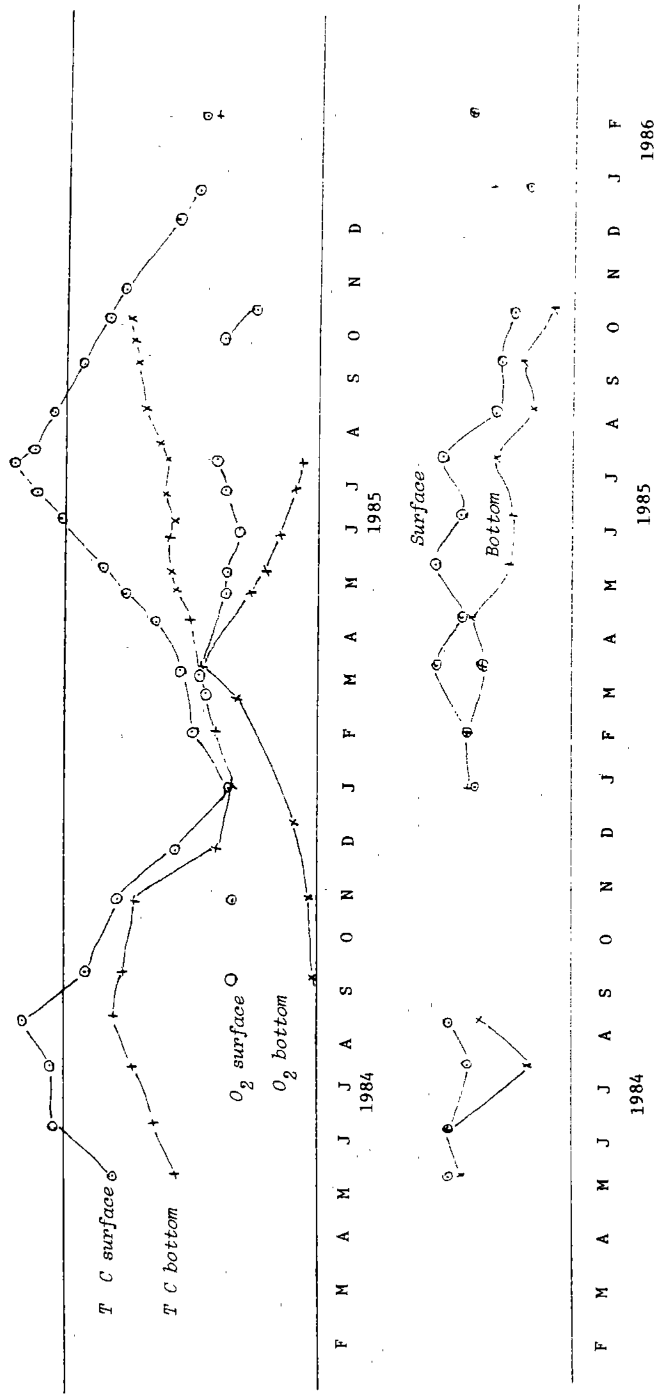
These high concentrations of suspended matter entail, among other things, overestimates of the total phosphorus concentrations in the water since an important fraction of the phosphorus is bound on the surfaces of the mineral particles [2].

The dissolved oxygen concentrations in the epilimnion vary between 9 and 12.5 mg/1. As shown in Figure 14, the oxygen concentrations in the epilimnion reflect the variations in the inflows during the winter and in the primary production during the summer.

The oxygen concentrations in the hypolimnion depend on the mixing period, the nitrate concentrations, the floods and the sedimentation of detritus. During the 1985 sampling, the dissolved oxygen was never exhausted in the hypolimnion. The nitrate reserves were used in the denitrification process, and no hydrogen sulfide was thus formed.

The phosphorus concentrations reflect the water inflows during the flood periods. One thus finds large variations in the total phosphorus concentrations during the winter periods (between 5 and 200 ug/1 in the reservoir). The variations in

Fig. 14. Temperature, dissolved oxygen and pH at the station close to the dam, 1984-1985



the phosphorus concentrations are much less important during the stratified periods (i.e. during the production season). The phosphorus concentrations in the epilimnion during this period vary between 10 and 57 ug/l with a mean value around 35 ug/l.

In spite of the important phosphorus charges from the Béja River, the phosphorus concentrations in the reservoir remain fairly low. This is probably due to a precipitation of phosphate by clay particles or by calcium ions, which entails a transformation of the phosphate to insoluble forms followed by a sedimentation but of the productive zone.

Figures 15 and 16 show the evolution of some physical and chemical parameters in the trophogenic zone during periods of phytoplankton production. In the hypolimnion, first the dissolved oxygen and then the nitrate are produced in the epilimnion by the microorganisms during the decomposition of the organic material; the nitrates serve as a supplementary source of oxygen that retards the use of sulfate oxygen in the decomposition process. The presence of nitrate thus retards or prevents the formation of hydrogen sulfide in the reservoir and the ensuing degradation of the water quality.

4. CHLOROPHYLL AND THE TROPHIC STATE

Chlorophyll concentrations have been determined and used as an indicator of the phytoplankton biomass in the reservoir since 1982.

The following (arithmetic) mean values in ug/l for the chlorophyll concentrations have been obtained for 1983-1985 at station No. 1 (close to the dam): 1983, 6.3; 1984, 3.9 and 1985, 4.3. The maximum values registered during the same period (at the same station) were, 19, 14 and 17 ug/l for 1983, 1984 and 1985, respectively.

Following the limiting values proposed in the OECD eutrophication study [3] for classifying the trophic state of a lake, the chlorophyll concentrations found in Sidi Salem lead to a classification of this reservoir as mesotrophic.

On the other hand, if the transparency or the total phosphorus concentrations are used, Sidi Salem becomes classified as eutrophic or on the limit between eutrophic and mesotrophic. However, neither the mean phosphorus concentrations nor the Secchi disc depths are directly related to the primary production or to the phytoplankton biomass because of the high content of mineral particles in the water. Consequently, chlorophyll concentrations are a more reasonable method for determining the trophic state of the reservoir.

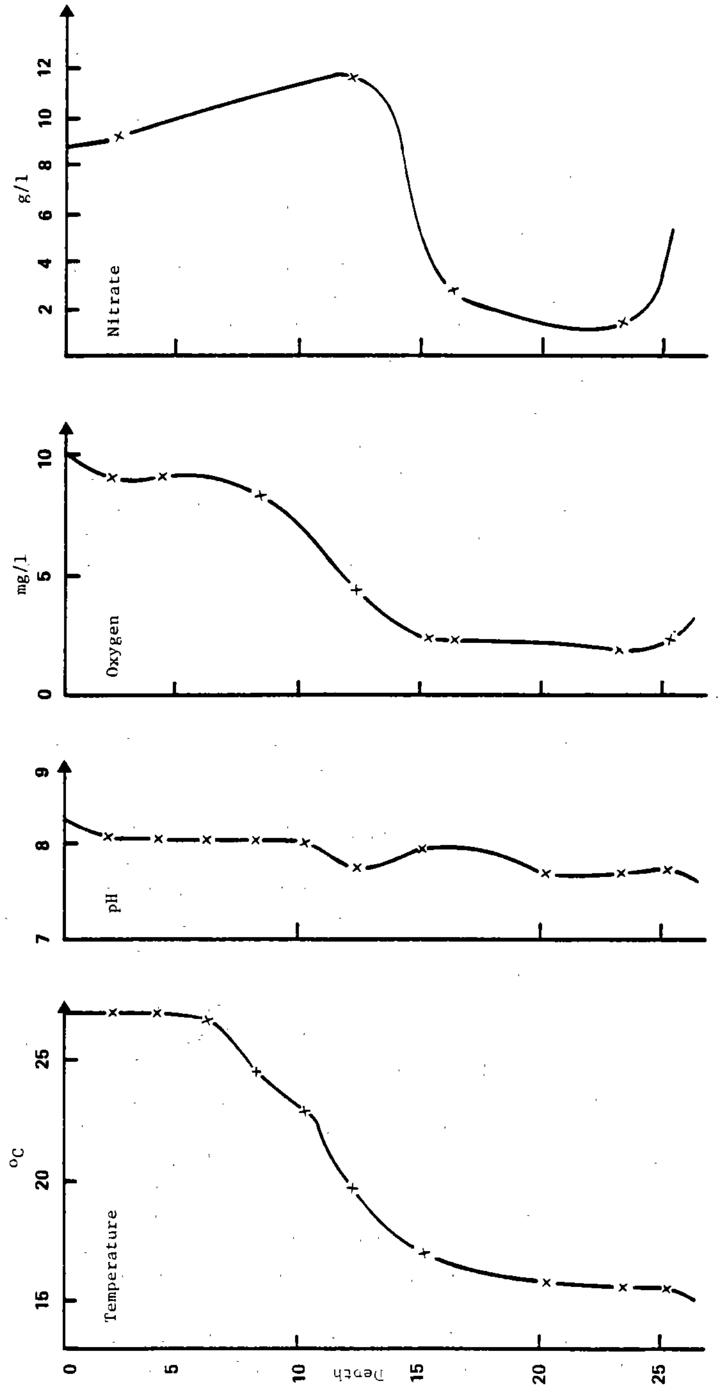


Fig. 15. *Stratification of temperature, pH, dissolved oxygen and nitrate, 22 July 1985*

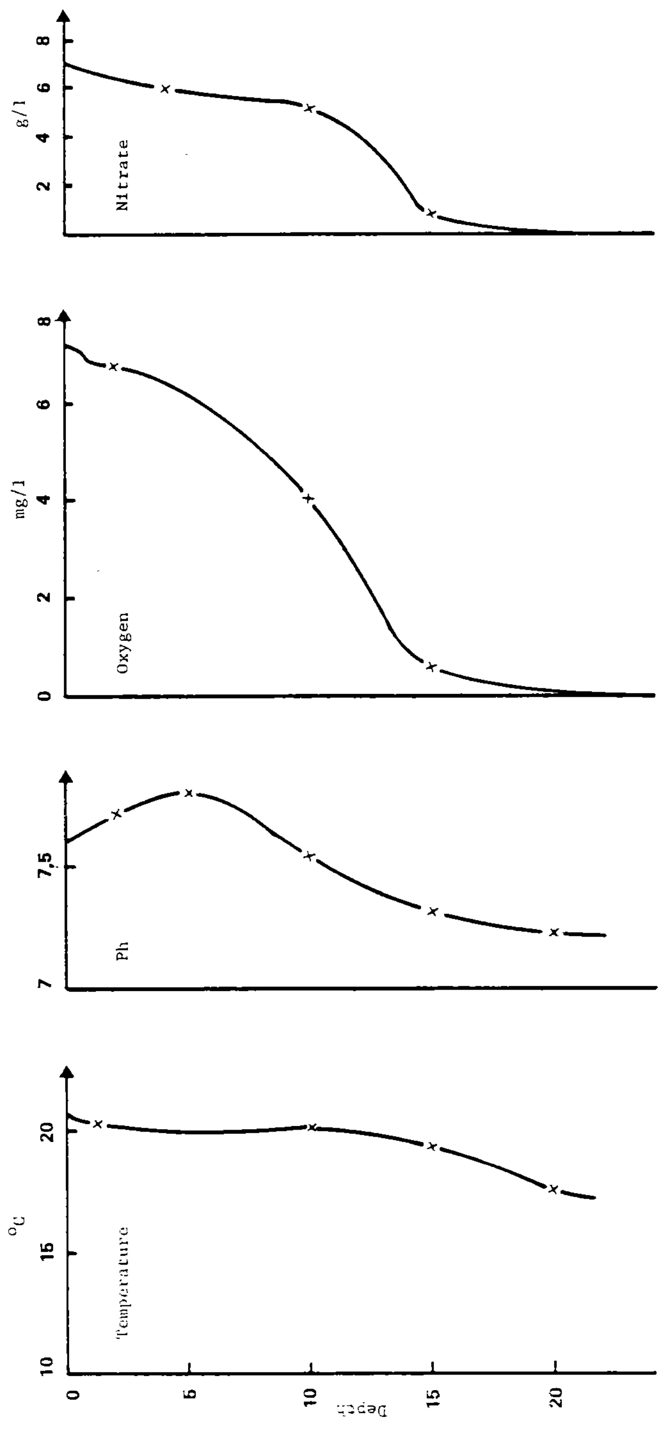


Fig. 16. Stratification of temperature, pH, dissolved oxygen and nitrate, 25 October 1985

5. ESTIMATIONS OF PHOSPHORUS LOAD AND ITS CONSEQUENCES

The report by Bernhardt [4] contains an estimate of the total quantity of phosphorus entering the reservoir from the drainage basin. This estimate was based on data available at that time, including data from other reservoirs. The loads were estimated according to their origin (i.e. domestic or industrial point sources and diffuse agricultural sources).

An effort to measure directly the phosphorus charge was initiated in 1979. This study, which lasted from December 1979 and November 1980, included contributions from the Medjerdah River and Béja River but not the Zarga River.

The preliminary estimates made by Bernhardt yielded an annual phosphorus charge of 70 tons, while the results of the measurements indicate that the annual charge should be between 40 and 50 tons. Table 7 gives estimates for the phosphorus loads per unit surface on the reservoir.

A comparison between measured and tolerable values for a mesotrophic or slightly eutrophic lake shows that the measured loads are much higher than the tolerable loads, meaning that the risk for Sidi Salem to become eutrophied would be very high if the phosphorus entering the reservoir were available to the algae. In supposing a hydraulic retention time of 0.5 y and a mean depth of 10 m, the application of the Vollenweider diagrams leads to the conclusion that the phosphorus loads are vastly excessive, yielding expected chlorophyll concentrations of about 40 ug/l.

Table 7. *Phosphorous loads in the Sidi Salem reservoir*

Level (m AMSL)	Surface (ha)	Phosphorus load from Bernhardt's estimates ($g/m^2/y$)	Phosphorus load from the 1979-1980 study ($g/m^2/y$)	Tolerable load ($g/m^2/y$)
100	2500	2.8	1.6-2.0	Mesotrophic: 0.2-0.4
105	3000	2.3	1.3-1.5	Slightly eutrophic: 0.4-0.6
110	4250	1.6	0.9-1.2	

In reality, however, the mean concentrations of chlorophyll are 4-6 ug/l and the reservoir can be classified as mesotrophic. Thus, the precipitation of phosphate, the turbidity of the water and the uneven yearly distribution of the phosphorus load must be taken into consideration in a eutrophication study of Sidi Salem.

REFERENCES

1. Krause, P. Evolution de la situation trophique dans la retenue de Side Salem. [Evolution of the trophic situation in the Sidi Salem reservoir]. Tunis, Ministry of Agriculture, 1984.

2. Lemoalle, J. Etude des conditions de milieu dans la retenue Sidi Salem. [Study on the environmental conditions in the Sidi Salem reservoir]. Paris, French Institute of Scientific Research for Development and Cooperation, 1985.

3. Eutrophisation des eaux. Méthodes de surveillance, d'évaluation et de lutte. [Eutrophication of water. Methods of surveillance, evaluation and strategy]. Paris, Organisation for Economic Co-operation and Development, 1982.

4. Bernhardt, H. Prognose zur Wasserqualität in der Talsperre Sidi Salem, Tunis. [Prognoses of the water quality in the Sidi Salem reservoir]. Tunis, Ministry of Agriculture, 1978.

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