Policy Analysis of Water Management for the Netherlands

Vol. VI, Design of Eutrophication Control Strategies

F.J. Los, N. M. de Rooij, J. G. C. Smits, J. H. Bigelow

February 1982

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The Netherlands Rijkswaterstaat

Rand
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DELFHYDRAULICS LABORATORY
The Netherlands
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The Netherlands Rijkswaterstaat
PREFACE

For some time the Netherlands has had a problem with water quality, particularly salinity, eutrophication, and thermal pollution. Moreover, the future demand for fresh water is expected to exceed the supply. The growing demand for the limited supply of groundwater is leading to increased competition among its users: agriculture, industry, nature preserves, and companies that supply drinking water. The supply of surface water is sufficient except in dry years, when there is competition not only among such users as agriculture, power plants, and shipping, but also among different regions.

Facing such water management problems, the Dutch government wanted an analysis to help draft the first national water management law and to select the overall water management policy for the Netherlands. It established the Policy Analysis for the Water Management of the Netherlands (PAWN) Project in August 1976 as a joint research project of Rand (a nonprofit corporation), the Rijkswaterstaat (the government agency responsible for water control and public works), and the Delft Hydraulics Laboratory (a leading Dutch research organization).

The primary tasks of the PAWN project were to:

1. Develop a methodology for assessing the multiple consequences of water management policies.
2. Apply it to develop alternative water management policies for the Netherlands and to assess and compare their consequences.
3. Create a Dutch capability for further such analyses by training Dutch analysts and by documenting and transferring methodology developed at Rand to the Netherlands.

The methodology and results of the PAWN project are described in a series of publications entitled Policy Analysis of Water Management for the Netherlands. The series contains the following volumes:

- Volume I, Summary Report (Rand R-2500/1)
- Volume II, Screening of Technical and Managerial Tactics (Rand N-1500/2)
- Volume III, Screening of Eutrophication Control Tactics (Rand N-1500/3)
- Volume IV, Design of Long-Run Pricing and Regulation Strategies (Rand N-1500/4)
- Volume V, Design of Managerial Strategies (Rand N-1500/5)
- Volume VA, Methodological Appendixes to Vol. V (Rand N-1500/5A)
-iv-

- Volume VI, Design of Eutrophication Control Strategies (Rand N-1500/6)
- Volume VII, Assessment of Impacts on Drinking-Water Companies and Their Customers (Rand N-1500/7)
- Volume VIII, Assessment of Impacts on Industrial Firms (Rand N-1500/8)
- Volume IX, Assessment of Impacts on Shipping and Lock Operation (Rand N-1500/9)
- Volume X, Distribution of Monetary Benefits and Costs (Rand N-1500/10)
- Volume XI, Water Distribution Model (Rand N-1500/11)
- Volume XII, Model for Regional Hydrology, Agricultural Water Demands and Damages from Drought and Salinity (Rand N-1500/12)
- Volume XIII, Models for Sprinkler Irrigation System Design, Cost, and Operation (Rand N-1500/13)
- Volume XIV, Optimal Distribution of Agricultural Irrigation Systems (Rand N-1500/14)
- Volume XV, Electric Power Reallocation and Cost Model (Rand N-1500/15)
- Volume XVI, Costs for Infrastructure Tactics (Rand N-1500/16)
- Volume XVII, Flood Safety Model for the IJssel Lakes (Rand N-1500/17)
- Volume XVIII, Sedimentation and Dredging Cost Models (Rand N-1500/18)
- Volume XIX, Models for Salt Intrusion in the Rhine Delta (Rand N-1500/19)
- Volume XX, Industry Response Simulation Model (Rand N-1500/20)

Four comments about this series of publications seem appropriate.
First, the series represents a joint Rand/Rijkswaternaart/Delft Hydraulics Laboratory research effort. Whereas only some of the volumes list Dutch coauthors, all have Dutch contributors, as can be seen from the acknowledgments pages.

Second, except where noted, these publications describe the methodology and results presented at the final PAWN briefing at Delft on December 11 and 12, 1979. For Rand, this briefing marked the beginning of the documentation phase of the project and the end of the analysis phase. Rand and the Rijkswaterstaat (RWS) considered the results to be tentative because (1) some of the methodology had not become available until late in the analysis phase, and (2) the RWS planned to do additional analysis.

Third, the RWS is preparing its Nota Waterhuishouding, the new policy document on water management scheduled for publication in 1982, by combining some of the PAWN results from December 1979 with the results of considerable additional analysis done in the Netherlands with the PAWN methodology. Because the understanding gained in the original analysis led to improvements in the data—and, in some instances, the models—used to represent the water management system in the additional analysis, the reader is hereby cautioned that the numerical results and conclusions presented in the PAWN volumes will not
always agree with those presented in the Nota Waterhuishouding or its companion reports. (It has not been possible to indicate such differences in the volumes since they are being published before the Nota.) Thus, the present series of publications puts primary emphasis on documenting the methodology rather than on describing the policy results.

Fourth, Vols. II through XX are not intended to stand alone, and should be read in conjunction with the Summary Report (Vol. I), which contains most of the contextual and evaluative material.

The present volume discusses the methodology and results of the special PAWN study of eutrophication control. It should be read in conjunction with Vol. III, Screening of Eutrophication Control Tactics, where most of the discussion of the cost and technical aspects of potential control tactics occurs. The present volume estimates the effectiveness of many of the tactics discussed in Vol. III for controlling phytoplankton blooms in the particular circumstances found in Dutch lakes.

This volume should be of considerable interest to those concerned with water quality problems in general, and with eutrophication in particular.

NOTES

1. Rand had had extensive experience with similar kinds of analysis and had been working with the Rijkswaterstaat for several years on other problems.

2. The Rand contract was officially with the Rijkswaterstaat, Directie Waterhuishouding en Waterbeweging (Directorate for Water Management and Water Movement), but numerous other parts of the Rijkswaterstaat contributed to the analysis.

3. Delft Hydraulics Laboratory research was performed under project number R1230, sponsored by the Netherlands Rijkswaterstaat.

4. Each water management policy involved a mix of tactics, each a particular action to affect water management, such as building a particular canal or taxing a particular use. Four kinds of tactics were considered: building new water management facilities (infrastructure) or applying various treatments to the water (called technical tactics); using managerial measures (called managerial tactics) to change the distribution of water among competing regions and users; and imposing taxes or quotas to affect the quantity or quality of water extracted or discharged by different users (called price and regulation tactics, respectively). A mix of tactics of the same kind is called a strategy. Thus, the overall policy could be conceived as a combination of technical, managerial, pricing, and regulation strategies.

5. Although the technical meaning of eutrophication is broader, the word as used here refers to large phytoplankton (i.e., algae) blooms. A propensity for large phytoplankton blooms is indeed characteristic of eutrophic waters, and is one of their most objectionable features.
SUMMARY

S.1. INTRODUCTION

Our work has suggested that strategies to improve water quality should be tailored to individual localities. To require uniformity in all parts of the Netherlands will result in strategies with limited effectiveness, high cost, or both. We have carried out a study of eutrophication, which is perhaps the most widespread and pressing water quality problem in the Netherlands; and we offer the study as an example of what is needed to determine promising local strategies to improve water quality.

In the Netherlands, eutrophication is taken to mean excessive growth of phytoplankton. Such growth can cause a number of problems. The oxygen concentration in the water can fluctuate, as the phytoplankton photosynthesize by day and respire by night. The collapse of the phytoplankton bloom (i.e., the sudden death of the phytoplankton) leaves a large amount of organic matter in the water, whose decomposition can deplete the water of oxygen and cause bad odors and taste. Some phytoplankton, particularly blue-greens, have a toxic effect on domestic animals. The growth of blue-green phytoplankton also results in turbid or scum-covered water. Finally, phytoplankton can interfere with water treatment, by clogging filters and intakes.

S.2. METHODOLOGY

We used three methodology components in the eutrophication study. Two of them, the nutrient model CHARON and the phytoplankton model BLOOM II are being developed under a study called WABASIM, which is a joint project of the Environmental Division of the Deltadienst (a branch of the RWS) and the Environmental Hydraulics Branch of the Delft Hydraulics Laboratory, with technical assistance from Rand. The third methodology component, the dissolved oxygen model OXYMOD, was developed under PAWN. These three components can be used separately, as was done for most of our analysis, or connected together as illustrated in Fig. S.1.

S.2.1. The Nutrient Model CHARON

In PAWN, we have not studied nutrients for their own sake, but only for the effect they have on phytoplankton blooms. Phytoplankton need nutrients (especially phosphorus, nitrogen, and for some species, silicon) in order to grow. Indeed, the tactic most often suggested for controlling phytoplankton blooms, and the primary tactic presently being considered for Dutch waters, is phosphorus reduction. The suggested means is to reduce the discharges of phosphorus into the surface water system, and thereby reduce the phosphorus load on eutrophic lakes.
Fig. S.1--Relations among eutrophication methodology components (inputs to these models from sources other than these models are not shown)
In deep lakes (e.g., 10 m), which are thermally stratified during the summer, the colder hypolimnion acts as a nutrient trap. As phytoplankton die, they sink below the thermocline (the upper boundary of the hypolimnion) carrying their load of nutrients beyond the reach of their living brethren. But most Dutch waters are far too shallow to become stratified, and there is evidence that the bottom sediment and interstitial pore water do not act as a simple nutrient trap. Thus, the simple nutrient models suitable for deep lakes are inadequate for the Dutch situation.

Accordingly, we constructed CHARON. It is a chemical model which includes both fast reactions, which we assume always to have progressed to equilibrium, and slow reactions, which need not have done so. Gas transfers between water and air, and nutrient transfers between the bottom and the bulk water, are slow reactions. So too is nutrient uptake or release by phytoplankton (this information is not calculated by CHARON, but must be specified either from observations or from BLOOM II outputs). Most chemical reactions within the bulk water are considered fast.

Calibration of the model to the excellent data from Grote Rug (a reservoir near Dordrecht) showed that the bottom indeed served as a steady source of nutrients. A study of these data also showed that in some years an "explosive" release of nutrients from the bottom occurred. This work confirmed our fears that reductions in phosphorus loadings might have a severely limited effect on phytoplankton blooms until such time as the nutrient-rich bottom sediments of Dutch lakes have been depleted of most of their nutrients.

S.2.2. The Phytoplankton Model BLOOM II

The phytoplankton model uses linear programming techniques to find the maximum potential phytoplankton biomass consistent with the prevailing abiotic conditions. The conditions that are included in the model as potential bloom-limiting factors are the three nutrients, phosphorus, nitrogen, and silicon, as well as solar energy (which is needed for photosynthesis). BLOOM II can obtain estimates of available nutrients either from observations or from CHARON. In its present configuration, the model can select among ten different species of phytoplankton, with different nutrient and energy requirements, for that combination which can most effectively use the available quantities of these factors.

Because factors not included in the model may sometimes be limiting, or because the growth history of a bloom may result in a different combination of species than that predicted by the model, BLOOM II will tend to overstate the risk of large blooms. However, underprediction is also possible due to uncertainties in some parameters, notably the phytoplankton mortality rate.
S.2.3. The Dissolved Oxygen Model OXYMOD

As mentioned earlier, a large phytoplankton bloom can cause diurnal fluctuations in the oxygen content of the water, and a collapse of the bloom can result in total oxygen depletion. To assess the risk of these phenomena, we developed a model of the dissolved oxygen budget of a lake. Among the terms included in the budget are respiration and photosynthesis by phytoplankton, mineralization of dead phytoplankton, mineralization of other organic matter, and reaeration (transfer of oxygen from the atmosphere to the water). This model takes as input the phytoplankton bloom in each week of a year, obtained either from direct observation or from BLOOM II predictions, and calculates both the diurnal oxygen fluctuation and the minimum oxygen level that would be achieved, week by week, if the bloom were to collapse suddenly.

S.3. LAKES SELECTED FOR STUDY

We have carried out this study for eleven Dutch lakes, which can be seen in Fig. S.2. Six of the lakes are among the IJssel lakes: the IJsselmeer, Markermeer, Veluwemeer, Wolderwijd, Gooimeer plus Eemmeer, and IJmeer. The Slotermeer is in the province of Friesland, just to the east of the IJsselmeer. The Westeinderplassen is a lake in the Midwest formed by peat excavations. The Haringvliet is in the Lower Rivers area. The Stuwpond Lith is the "lake" formed by the water impounded above the weir in the Maas at Lith. It is a lake during the summer, but during winter, when the Maas flow becomes large, it is merely another part of the river. Finally, we have considered the future Kommeer, which will be the shallow part of the future Zoommeer that lies near Bergen op Zoom.

The most important descriptors of these lakes are their average depths and their background extinction coefficients. The background extinction coefficient is the fraction of incident light absorbed by nonphytoplankton substances for each meter that the light penetrates. Light absorbed by such substances is not available for phytoplankton. Table S.1 shows these two quantities for each of the eleven lakes studied.

The details of eutrophication in the Netherlands are highly time and site specific. Using BLOOM II and data found in the literature to investigate the present situation in these lakes, we found that different biomass concentrations, limiting factors, and bloom compositions are expected in different lakes and for different parts of the year. For each of the main constraints (nitrogen, phosphorus, and energy), there is at least one lake in which it is more important than any other factor. But for the cases considered here, nitrogen and energy are more frequently limiting than phosphorus. Also the computed yearly maxima (almost always during summer) are usually nitrogen or energy, rather than phosphorus, limited.
Fig. S.2--Lakes investigated in PAWN's eutrophication study
Table S.1
DEPTH AND BACKGROUND EXTINCTION COEFFICIENTS FOR LAKES IN EUTROPHICATION STUDY

<table>
<thead>
<tr>
<th>Lake</th>
<th>Depth (m)</th>
<th>Background Extinction Coefficient (l/m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>IJsselmeer</td>
<td>4.5</td>
<td>1.8</td>
</tr>
<tr>
<td>Markermeer</td>
<td>3.3</td>
<td>4.0 (a)</td>
</tr>
<tr>
<td>Veluwemeer</td>
<td>1.2</td>
<td>4.5</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>1.5</td>
<td>2.5</td>
</tr>
<tr>
<td>Gooi/Eemmeer</td>
<td>1.7</td>
<td>5.5</td>
</tr>
<tr>
<td>IJmeer</td>
<td>2.0</td>
<td>2.5 (a)</td>
</tr>
<tr>
<td>Slotermeer</td>
<td>1.2</td>
<td>6.5</td>
</tr>
<tr>
<td>Westeinder</td>
<td>2.5</td>
<td>4.0</td>
</tr>
<tr>
<td>Haringvliet</td>
<td>6.0</td>
<td>1.2</td>
</tr>
<tr>
<td>Stuwpend Lith</td>
<td>5.0</td>
<td>2.3</td>
</tr>
<tr>
<td>Zoommeer</td>
<td>2.5</td>
<td>1.2</td>
</tr>
</tbody>
</table>

(a) For these lakes, the background extinction coefficient was highly variable. We estimated different values in different weeks of the year based on an observed correlation with wind speed. The values in the table are typical of the weekly values actually used.

S.4. EUTROPHICATION MEASUREMENT AND STANDARDS

S.4.1. Nutrient (Phosphate) Standards

Phytoplankton require nutrients in order to grow. (The three we consider in BLOOM II are phosphate, nitrogen, and silicon.) This suggests that controlling nutrients will indirectly control phytoplankton. Accordingly, Ref. S.1 suggests water quality standards for nutrients. The most important of these for our purposes are the standards for phosphate, because they have been proposed for the sole purpose of controlling phytoplankton blooms, and because the Dutch have begun implementing a program to reduce phosphate discharges into their surface waters. The standards suggested in Ref. S.1 are of two kinds, a provisional limit and a stricter target value. For phosphate, the provisional limit is 0.3 mg/l, and the target value 0.05 mg/l.¹
S.4.2. Chlorophyll Standards

The amount of phytoplankton is usually measured in terms of the chlorophyll concentration, since instruments are available that make such measurements simple. The Rijksinstituut voor Zuivering van Afvalwater (State Institute for Wastewater Treatment, RIZA) has therefore expressed their suggested water quality standards as maximum allowable chlorophyll concentrations, including a lenient standard of 100 ug/l (micrograms per liter) and a strict one of 50 ug/l. However, the problems caused by phytoplankton correlate more closely with the biomass, expressed as dry weight. The ratio of dry weight to chlorophyll content varies over a factor of two, depending on the size and species of the phytoplankton cells present. BLOOM II predicts the biomass of phytoplankton present, and not the chlorophyll. Thus, in our study each of RIZA's standards is expressed as a range of biomass, whose upper and lower limits differ by a factor of two.

S.4.3. Dissolved Oxygen Standards

Perhaps the most objectionable aspect of phytoplankton blooms is the large mass of organic matter left behind when the phytoplankton die. The dead cells sink to the bottom, and must be mineralized by bacterial action. This imposes a demand for oxygen on the water near the bottom, and can result in severe oxygen depletion and sometimes in anaerobic conditions. In Ref. S.1, the official Dutch five-year plan for combating surface water pollution, a provisional range (rather than a single limit) and a target range for dissolved oxygen are proposed. Provisionally, dissolved oxygen should be no less than 50 percent of saturation and no more than 150 percent of saturation. The target range is between 80 and 120 percent of saturation. The ranges are expressed in terms of percent of saturation because the amount of oxygen present at saturation varies strongly with temperature, and hence even 100-percent oxygen-saturated water (evidently considered the ideal in Ref. S.1) would sometimes violate a standard expressed in terms of amounts. In addition, the percent of saturation is a good measure of the ability of aquatic life to extract oxygen from the water.

S.5. TACTICS CONSIDERED IN BLOOM II SIMULATIONS

We considered six different tactics in BLOOM II simulations which had been suggested as possibly effective (at least under some conditions) in reducing blooms of either all phytoplankton or of blue-green phytoplankton only. They are:

1. Addition of silicon, a nutrient required by some phytoplankton species.
2. Reduction of phosphate, a nutrient required by all phytoplankton.
3. Increasing the background extinction coefficient.
5. Flushing.
6. Dredging.

These tactics are generic, in that we have only tentative suggestions in mind for implementing them. For example, reducing phosphate can be accomplished by removing it from treated sewage rather than allowing it to remain in the treated discharge, by precipitating phosphate by means of iron or aluminum compounds added to the lake or reservoir, or possibly by other means as well. In addition, a particular implementation may have elements of several of our generic tactics. Dredging, for example, not only increases the depth but removes nutrients sequestered in the bottom sediments.

S.5.1. Silicon Addition

The tactic of silicon addition is an attempt to defeat blue-green phytoplankton, which are the most objectionable species, by other species. Objections to blue-greens are partly due to the fact that few fish can eat blue-greens, while many feast on other species. Thus, blue-greens depress the value of a lake for commercial and recreational fishing. Also, blue-greens "clump" together, forming an unsightly scum on the water surface, whereas other species distribute themselves almost invisibly throughout the water.

Silicon addition tries to promote the growth of diatoms in place of blue-greens. Diatoms, however, require a plentiful supply of silicon which they incorporate in their microscopic shells. If the only factor restraining the growth of diatoms were the lack of silicon, then adding silicon would cause the diatoms to grow, using up not only silicon but all the other nutrients as well. The other nutrients (e.g., phosphate, nitrogen) are also needed by blue-greens, so the growth of diatoms could reduce the blue-green population.

Unfortunately, the model results suggest that silicon addition is ineffective under almost all circumstances. Blue-greens are more efficient users of light and nutrients than diatoms are, and outdo the diatoms even in the presence of ample silicon.

S.5.2. Phosphate Reduction

The most widely suggested tactic for controlling phytoplankton blooms is reduction of phosphate. It is guaranteed to work if the phosphate is reduced to sufficiently low levels, because every cell of phytoplankton requires a certain minimum amount of phosphate. To simulate this tactic in BLOOM II, we parametrically reduced the available phosphate concentration, holding all other inputs (e.g., other nutrients, light intensity) constant.
Figure 8.3 shows the results for our eleven test lakes. For each lake there is a line in the figure with a dot at its right-hand end. The dot is positioned horizontally at the maximum total phosphate concentration observed in the lake in 1976. The total phosphate is the sum of the dissolved phosphate and the phosphate incorporated in phytoplankton. The dot is positioned vertically at the maximum phytoplankton biomass calculated by the model for that lake in 1976. The fact that the upper right-hand section of each line is horizontal indicates that the phytoplankton in the maximum calculated bloom do not contain the maximum total phosphate observed during the year. It is necessary to reduce the phosphate below the amount incorporated in the maximum bloom before phosphate reduction can affect the bloom size.

Several observations may be made from this figure. First, in the lakes with the largest blooms, enormous percentage phosphate reductions are necessary before the RIZA chlorophyll standards are met. (The figure shows the ranges of biomass which correspond to the 100 ug/l and 50 ug/l chlorophyll standards. Recall that there is an uncertainty of a factor of two in the biomass that corresponds to any chlorophyll level.) Second, the provisional limit of 0.3 mg/l of phosphate is far too lenient to meet even the more lenient chlorophyll standard (again, see Note 1). Finally, the target value of 0.05 mg/l of phosphate may be (or may not) be strict enough to meet the 50-ug/l-chlorophyll standard.

There now arises the question, What must be done to achieve a reduction of "X" percent in the phosphate concentration? According to our analysis with the nutrient model CHARON, there is a steady flux of phosphate from the bottom to the bulk water. At the phosphate concentrations presently found in most Dutch lakes, the net flux is out of the water and into the bottom, but if the phosphate concentration in the water were reduced, the net flux might be outward. In addition, explosive fluxes of phosphate from the bottom can occur. Our studies have indicated explosive fluxes occurring in the IJsselmeer, Velpemeer, Wolderwijd, and the Gooi/Eemmeer in 1976, and in the Veluwemeer and Wolderwijd in 1975. Other instances may also have occurred; results were ambiguous.

To reduce phosphate concentrations, both the contributions from inflowing waters and the net flux from the bottom must be controlled. Phosphate can be removed from point sources--the Dutch have a program to do so [S.1]--but much phosphate will remain in some waters (e.g., the Rijn). There are also techniques for sealing lake bottoms to prevent or slow nutrient release, but many of these techniques may harm bottom organisms (an undesirable side effect). In addition, the Dutch have been experimenting for several years with precipitating phosphate from the inlet water of a reservoir (the Grote Rug) using iron and aluminum compounds. Preliminary calculations and the results from Grote Rug agree that the phosphate reductions necessary to meet RIZA's chlorophyll standards will not come easily.
Fig. S.3—Effect of phosphate reduction on yearly maximum biomass predictions
S.5.3. Increasing Background Extinction

Phytoplankton must absorb light to live. In principle, if the background extinction coefficient were large enough, so little light would be left for phytoplankton that only a few cells could survive. Special colored dyes exist that absorb light when dissolved in water. They can even be tailored to absorb only those wavelengths of light that phytoplankton exploit most. A rough estimate of their cost is $0.0046 \text{Dfl/m}^2/\text{yr}$ (Vol. III). In our discussions with the Dutch, we found considerable resistance to this tactic on aesthetic grounds. In any case, the model results shown in Fig. S.4 suggest that this tactic is effective only in deep lakes, in which phytoplankton blooms are generally not a problem.

S.5.4. Complete Mixing

One of the advantages that blue-greens have over competing species is an ability to control their vertical position in the water column. By adjusting their buoyancy, they occupy the depths at which the light intensity and nutrient concentrations are most favorable for growth. This ability can be overcome, however, by gently mixing the water column, for example by pumping air through a perforated pipe on the bottom and letting the rising bubbles stir the water. This method of mixing has been tried at the Biesbosch reservoirs near Rotterdam [S.2] with mixed but promising results.

In BLOOM II, each species of phytoplankton is assigned a depth in which to grow. For most species, the depth is the same as the actual depth of the lake; but for blue-greens it is less than the lake depth (the assigned depth being determined empirically), in order to represent their talent for controlling their vertical position. To represent the complete mixing tactic, we have made the depth for blue-greens equal to the lake depth.

As may be seen in Table S.2, complete mixing effectively reduces the maximum total biomass in lakes which have either a large background extinction, a large depth, or both. In the deep Markermeer, the predicted maximum bloom level is strongly reduced compared to the nominal case. In the IJsselmeer, which although deep has a low background extinction, and in the Gooimeer, Slotermeer, and Westeinder, which have low depths but high background extinctions, maximum total biomass is reduced by about one-third to one-half. In the nominal cases, maximum blooms are strongly dominated by blue-green phytoplankton (except in the Zoommeer). Complete mixing reduces blue-greens substantially in all lakes, although they remain dominant in some of them.
Fig. S.4--Effect of increasing the background extinction coefficient on yearly maximum biomass predictions.
Table S.2
EFFECT OF COMPLETE MIXING ON YEARLY MAXIMUM
BIOMASS PREDICTIONS
(mg/m³)

<table>
<thead>
<tr>
<th>Lake</th>
<th>Nominal Simulations</th>
<th>Complete Mixing</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total Biomass</td>
<td>Blue-Green Biomass</td>
</tr>
<tr>
<td>IJsselmeer</td>
<td>22.6</td>
<td>22.6</td>
</tr>
<tr>
<td>Markermeer</td>
<td>21.6</td>
<td>21.6</td>
</tr>
<tr>
<td>Veluwemeer</td>
<td>47.4</td>
<td>47.4</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>34.4</td>
<td>34.4</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>50.1</td>
<td>50.1</td>
</tr>
<tr>
<td>IJmeer</td>
<td>30.4</td>
<td>30.4</td>
</tr>
<tr>
<td>Slotermeer</td>
<td>29.1</td>
<td>29.1</td>
</tr>
<tr>
<td>Westeinder</td>
<td>16.3</td>
<td>16.3</td>
</tr>
<tr>
<td>Zoommeer</td>
<td>45.5</td>
<td>39.0</td>
</tr>
</tbody>
</table>

S.5.5. Flushing

Flushing is the continuous replacement of water in the lake by water from outside. The water taken from the lake carries with it both live and dead phytoplankton, as well as nutrients dissolved in the water. It is by increasing the effective mortality of the phytoplankton and by getting rid of nutrients that the tactic can reduce blooms. However, the replacement water also contains nutrients, and depending on their concentrations, the tactic may do more harm than good.

To simulate this tactic in BLOOM II, we assume that the nutrient concentrations in the replacement water are just sufficient to maintain the total amount of each nutrient constant. Any other assumption would combine the flushing tactic with a nutrient-reducing (or increasing) tactic. We also assume that the replacement water contains only dissolved nutrients, and no live or dead phytoplankton.

The results are shown in Fig. S.5. In lakes deeper than 3 m, flushing produced the expected decrease in phytoplankton biomass. In shallow lakes, however, the yearly biomass maxima increase, rather than decrease, for all attainable flushing rates. The reason is that flushing has two contradictory effects. First, the phytoplankton have to devote more of their reproductive potential to replacing losses rather than to actual growth. But as long as they have the reserve potential to do so, they profit from flushing, since the dead phytoplankton are swept out, and the nutrients they contain are replaced in dissolved form in the replacement water. The net effect is to make more nutrients available to live phytoplankton.
Fig. S.5--Effect of flushing on yearly maximum biomass predictions
In deep lakes, the yearly bloom maxima are generally limited by the availability of light, which cannot penetrate far into the water. Energy is simply not available for phytoplankton to reproduce rapidly, so flushing at any rate reduces the bloom maxima. In shallow lakes, however, the bloom peaks are usually limited by the availability of nutrients, and a reserve reproductive potential exists. Thus, only if flushing occurs at very high rates will the reproductive reserve be depleted, and the bloom peak reduced.

S.5.6. Dredging

As Fig. S.6 shows, dredging to increase the depth of a lake proved to be a universally effective tactic for reducing bloom maxima when simulated by BLOOM II. However, it generally proved necessary to deepen lakes by several meters before RIZA's chlorophyll standards were met. Dredging until the standards are met involves moving and disposing of enormous quantities of spoils. For example, for the Veluwemeer alone, dredging to 5.5 m, where the 50-µg/l standard is certain to be met, requires the removal of about 280 million m³ of bottom sediment. We have estimated the cost of dredging in the Waal to be 16.2 Dfl/m³ (Vol. XVIII). If we use this cost for the Veluwemeer as well, we find the total cost of dredging to 5 m to be over 4500 Dflm. Although dredging costs are highly variable, we think this estimate is not orders of magnitude too high, and that for all but small lakes and reservoirs, dredging is therefore probably prohibitively expensive.

However, dredging to remove only the top 10 cm of bottom sediments has been suggested. This layer of sediment contains much of the nutrient sequestered in the bottom, and its removal could be an effective nutrient reduction tactic. There are technical problems associated with preventing the nutrient-laden interstitial water from flowing back into the lake from the dredging barge, and there are difficulties in disposing of the spoils. But removing only 10 cm of sediment from a lake the size of the Veluwemeer would cost only 100 Dflm (again using the cost of dredging we estimated for the Waal), and hence may not be prohibitively expensive.

S.6. DISSOLVED OXYGEN

In the lakes studied, phytoplankton dominate the dissolved oxygen budget. BOD from external sources is much less important. The provisional (lower) limit of 50-percent saturation for dissolved oxygen is violated in the most eutrophic lakes, namely Wolderwijd, Veluwemeer, and Gooi/Eemmeer. The target level (80-percent saturation) is violated in all the lakes.

Calculations have shown that most lakes will suffer from severe oxygen depletion problems upon the collapse of a bloom. These anaerobic conditions may last for several days. Complete anaerob y is
Fig. S.6--Effect of dredging on yearly maximum biomass predictions
unlikely in three lakes, namely Markermee, IJmeer, and Westeinderplassen, but even in these the dissolved oxygen concentrations can drop below 50-percent saturation.

A reduction in chlorophyll to the suggested RIZA standard of 50 ug/l or even 100 ug/l will generally be sufficient to prevent the dissolved oxygen concentration from dropping below the 50-percent-saturation provisional limit. In some lakes, a chlorophyll level as high as 60 ug/l may be sufficient to ensure that the dissolved oxygen target level of 80-percent saturation is met. However, chlorophyll must be reduced to 30 ug/l to guarantee that the target level for dissolved oxygen is met in all lakes. This range reflects the differences in the chlorophyll to biomass ratio and differences in local conditions, such as depth. A chlorophyll standard between 50 and 100 ug/l will ensure that a bloom collapse will not cause anaerobic conditions.

We have used OXYMOD to assess the impact of two eutrophication control tactics on the dissolved oxygen budget. The first tactic, phosphate reduction, has limited impact on the oxygen budget; reductions of up to 90 percent are needed to ensure that the target level of 80 percent saturation for dissolved oxygen is met. The second tactic, dredging, also has a limited impact. At a depth of 3 m or more, dissolved oxygen budgets can improve significantly, but the risk of anaerobic conditions following the collapse of a bloom remains high.

S.7. CONCLUSIONS AND RECOMMENDATIONS

PAWN's eutrophication study justifies three general conclusions and recommendations. First, by itself, the preferred Dutch eutrophication control strategy of phosphate reduction will be largely ineffective. The Dutch plan to control phosphate discharges into the surface water from point sources (mostly sewage treatment plants) within the Netherlands. But Rijn and Maas water from outside the Netherlands already contains high concentrations of phosphate. Further, the bottom sediments of eutrophic Dutch lakes are phosphate rich, and their phosphate is available to a significant degree to support phytoplankton blooms.

In fact, no single tactic is capable of solving the eutrophication problem in all lakes. Thus, overall reductions of phytoplankton blooms are possible only if the eutrophication control strategy is specially adapted to regional and local circumstances. For example, phosphate reduction could be employed for areas not receiving much Rijn or Maas water, increasing the background extinction could be tried in deep lakes, and dredging could be tried in small lakes and reservoirs.

Finally, we think a single tactic will seldom solve the problem in any lake, and that combinations of tactics should be considered. While we have not simulated any such combinations, we can suggest a few possibilities. For example, a reduction in phosphate loading
could be combined with dredging of the top few decimeters of the bottom sediments, which are rich in nutrients. Or an increase in depth by dredging could be combined with artificial mixing in reservoirs, which have to be dredged anyway. This latter combination would be more effective if the background extinction were relatively high as well.

NOTE

1. The provisional phosphate limit of 0.3 mg/l is specified in Ref. S.1, which is known as the IMP for 1975-1979. The newer IMP for 1980-1984, which was not available in time for use in this study, lowers the provisional limit for phosphate to 0.2 mg/l.

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CONTENTS

PREFACE .......................................................................................................................... iii

SUMMARY ....................................................................................................................... vii

ACKNOWLEDGMENTS ................................................................................................. xxv

FIGURES .......................................................................................................................... xxxi

TABLES ............................................................................................................................. xxxiii

Chapter
1. GENERAL INTRODUCTION ....................................................................................... 1
   1.1. Eutrophication in the Netherlands ............................................................... 1
   1.2. WABASIM Project ....................................................................................... 2
   1.3. Use of WABASIM Models in PAWN ............................................................. 4
   1.4. Selection of PAWN Lakes ......................................................................... 6
   1.5. Phytoplankton ............................................................................................. 6
   1.6. Nutrient Cycles ......................................................................................... 9
   1.7. Dissolved Oxygen .................................................................................... 11
   1.8. Organization of This Volume ................................................................... 12
   Notes .................................................................................................................. 12
   References ........................................................................................................ 13

PART I: THE NUTRIENT MODEL (CHARON)

2. INTRODUCTION ........................................................................................................ 17

3. NUTRIENT PROCESSES ....................................................................................... 19
   3.1. Exchange Reactions with the Bottom ....................................................... 19
   3.2. Nutrient Reactions in Water .................................................................... 25
   3.3. Phytoplankton .......................................................................................... 33
   References ........................................................................................................ 33

4. FORMULATION OF THE NUTRIENT MODEL ................................................. 35
   4.1. The Equilibrium Module ........................................................................... 35
   4.2. The Slow Reaction Module ..................................................................... 42
   4.3. Components Selected ............................................................................... 48
   4.4. Chemical Species Selected ..................................................................... 48
   4.5. Processes Included in the Slow Reaction Module ..................................... 49
   4.6. Inputs to the Model .................................................................................. 49
   Notes ................................................................................................................ 50
   References ........................................................................................................ 50

5. CALIBRATION AND VALIDATION OF THE NUTRIENT MODEL .............. 51
   5.1. The Calibration Process .......................................................................... 51
   5.2. Calibration Results .................................................................................. 53
   5.3. Validation ................................................................................................ 62
   Reference ......................................................................................................... 66
## PART II: THE PHYTOPLANKTON MODEL (BLOOM II)

7. STRUCTURE OF THE PHYTOPLANKTON MODEL BLOOM II 75
   7.1. Purpose of BLOOM II 75
   7.2. Selection of Phytoplankton Species 76
   7.3. Nutrient Requirements of Phytoplankton Species 77
   7.4. Energy Requirements of Phytoplankton Species 81
   7.5. Solution Algorithm 95
   7.6. Behavior and Performance 96
   Notes 97
   References 98

8. DATA ACQUISITION FOR BLOOM II 99
   8.1. Inputs to the Model 99
   8.2. Data Availability for PAWN Lakes 101
   References 105

9. CALIBRATION AND VALIDATION 106
   9.1. Nominal Simulations with BLOOM II 106
   9.2. Model Outputs 106
   9.3. Some Results for GROTE RUG 107
   9.4. Validations (Nominal Results) for PAWN Lakes 110
   9.5. Fixation of Atmospheric Nitrogen 132
   Notes 133
   References 133

## PART III: THE DISSOLVED OXYGEN MODEL (OXYMOD)

10. INTRODUCTION TO THE OXYGEN MODEL 137
    Symbols Used for OXYMOD 138
    Reference 139

11. THE OXYGEN MODEL (OXYMOD) 140
    11.1. Objectives and Model Output 140
    11.2. The Worst Case Criterion 141
    11.3. Structure and Main Assumptions of OXYMOD 142
    11.4. The Mathematical Model 144
    References 153

12. CALIBRATION OF OXYMOD 154
    12.1. Calibration Setup 154
    12.2. Discussion of the Calibration Results 155
    Note 157
    References 158

13. DISSOLVED OXYGEN SIMULATIONS WITH OXYMOD 159
    13.1. Data Acquisition 159
    13.2. Discussion of Nominal Simulation Results 161
    13.3. Summary of Nominal Simulation Results 187
14. RECOMMENDATIONS FOR FUTURE RESEARCH .......... 189
  14.1. Improvement of the Dissolved Oxygen Model .... 189
  14.2. Improvement of Measurements of Calibration Data . 190
  14.3. Improvements in Predicting Impacts of Tactics ... 190

15. CONCLUSIONS FROM OXYMOD VALIDATION RUNS .......... 191

PART IV: RESULTS AND CONCLUSIONS

16. EVALUATION OF EUTROPHICATION CONTROL TACTICS .......... 195
  16.1. Limits to the Current Approach .................. 195
  16.2. Eutrophication Control Standards ................ 196
  16.3. Tactics Involving Available Nutrient Concentrations .................. 201
  16.4. Tactics Involving Available Energy ............... 204
  16.5. Impacts of Promising Tactics on Oxygen .......... 209
  16.6. Overview of Eutrophication Control Tactics ...... 210
  16.7. Some Alternative Recommendations to National Policies .......................... 212

Notes .............................................. 213
References ........................................ 213
FIGURES

S.1. Relations among eutrophication methodology components ...... viii
S.2. Lakes investigated in PAWN's eutrophication study .......... xi
S.3. Effect of phosphate reduction on yearly maximum biomass
predictions ............................................................ xvi
S.4. Effect of increasing the background extinction coefficient
on yearly maximum biomass predictions .......................... xviii
S.5. Effect of flushing on yearly maximum biomass predictions ... xx
S.6. Effect of dredging on yearly maximum biomass predictions ... xxii
1.1. Relations among eutrophication methodology components ...... 5
1.2. Lakes investigated in PAWN's eutrophication study .......... 7
1.3. Basic nutrient cycle used for phytoplankton and nutrient
models ................................................................. 10
5.1. Observed chlorophyll concentration in Ring 2, 1977 ........... 54
5.2. Measured and computed pH in Ring 2, 1977 ................... 55
5.3. Measured and computed O_2 in Ring 2, 1977 .................... 56
5.4. Measured and computed total P in Ring 2, 1977 ............... 57
5.5. Measured and computed nitrate in Ring 2, 1977 ............... 58
5.6. Computed nitrogen fluxes in Ring 2, 1977 ...................... 59
5.7. P fluxes in Ring 2, 1977, with measured loadings and
computed sedimentation .............................................. 60
5.8. Measured and computed Si in Ring 2, 1977 .................... 61
5.9. Measured and computed total P in Ring 3, 1976 ................ 63
5.10. Measured and computed Si in Ring 3, 1976 .................... 64
5.11. Measured and computed nitrate in Ring 3, 1976 ............... 65
6.1. Measured nitrate in IJsselmeer, 1976 ........................... 69
6.2. Measured Si in IJsselmeer, 1976 ............................... 70
6.3. Measured total P in IJsselmeer, 1976 ............................ 71
7.1. Photosynthetic efficiency curves of phytoplankton species
in the model (data derived from Grote Rug measurements by
the Delta Department) .................................................. 83
9.1. Grote Rug, Ring 2, 1977: Predicted and observed chlorophyll
in mg/m³; calibration run with measured mortality rates .... 108
9.2. Grote Rug, Ring 3, 1976: Predicted and observed chlorophyll
in mg/m³; calibration run ........................................... 109
9.3. IJsselmeer, 1976: Predicted and observed chlorophyll in
mg/m³; nominal run ................................................ 111
9.4. Markermeer, 1976: Predicted and observed chlorophyll in
mg/m³; nominal run ................................................ 113
9.5. Markermeer, 1976: Predicted and observed chlorophyll in
mg/m³; weekly background extinctions from wind speeds;
additional growth constraints ...................................... 114
9.6. Veluwemeer, 1975: Predicted and observed chlorophyll in
mg/m³; nominal run ................................................ 115
9.7. Veluwemeer, 1976: Predicted and observed chlorophyll
mg/m³; nominal run ................................................ 116
9.8. Wolderwijd, 1975: Predicted and observed chlorophyll in
mg/m³; nominal run ................................................ 118
9.9. Wolderwijd, 1976: Predicted and observed chlorophyll in mg/m²; nominal run .............................................. 119
9.10. Gooimeer, 1976: Predicted and observed chlorophyll in mg/m²; nominal run .................................................. 120
9.11. IJmeer, 1976: Predicted and observed chlorophyll in mg/m²; nominal run ....................................................... 122
9.12. IJmeer, 1976: Predicted and observed chlorophyll in mg/m²; weekly background extinctions from wind speeds .......... 123
9.13. Slotermeer, 1976: Predicted and observed chlorophyll in mg/m²; nominal run ................................................... 124
9.14. Westeinderplassen, 1976: Predicted and observed chlorophyll in mg/m²; nominal run ......................................... 125
9.15. Haringvliet, 1976: Predicted and observed chlorophyll in mg/m²; nominal run .................................................. 126
9.16. Stuwpand Lith, 1976: Predicted and observed chlorophyll in mg/m²; nominal run ............................................. 128
9.17. Zoommeer, 1976: Predicted chlorophyll in mg/m²; inputs taken from Haringvliet ................................................. 129
9.18. Zoommeer, 1976: Predicted chlorophyll in mg/m²; all inputs taken from Haringvliet, but background extinction 2.5 m⁻¹ ........................................................................ 130

11.1. Block diagram of the dissolved oxygen model .................... 143

12.1. 0₂ concentration in Ring 3 of De Grote Rug in 1976:

Final calibration results ....................................................... 156

13.1. Simulation oxygen model, IJsselmeer, 1976 (case A) .............. 163
13.2. Simulation oxygen model, IJsselmeer, 1976 (case B) .............. 164
13.3. Simulation oxygen model, Markermeer, 1976 (case A) ............ 166
13.4. Simulation oxygen model, Markermeer, 1976 (case B) ............ 167
13.5. Simulation oxygen model, Veluwemeer, 1976 (case A) ............ 168
13.6. Simulation oxygen model, Veluwemeer, 1976 (case B) ............ 169
13.7. Simulation oxygen model, Wolderwijd, 1976 (case A) ............ 170
13.8. Simulation oxygen model, Wolderwijd, 1976 (case B) ............ 171
13.9. Simulation oxygen model, Gooimeer, 1976 (case A) .............. 173
13.10. Simulation oxygen model, Gooimeer, 1976 (case B) .............. 174
13.11. Simulation oxygen model, IJmeer, 1976 (case A) ................. 175
13.12. Simulation oxygen model, IJmeer, 1976 (case B) ................. 176
13.13. Simulation oxygen model, Slotermeer, 1976 (case A) ............ 177
13.15. Simulation oxygen model, Westeinderplassen, 1976 (case A) .. 179
13.16. Simulation oxygen model, Westeinderplassen, 1976 (case B) . 180
13.17. Simulation oxygen model, Haringvliet, 1976 (case A) ............ 182
13.18. Simulation oxygen model, Haringvliet, 1976 (case B) ............ 183
13.20. Simulation oxygen model, Stuwpand Lith, 1976 (case B) ....... 185
13.21. Simulation oxygen model, Zoommeer, 1976 (case B) ............ 186

16.1. Effect of phosphate reduction on yearly maximum biomass predictions .......................................................... 197
16.2. Effect of increasing the background extinction coefficient on yearly maximum biomass predictions ......................... 198
16.3. Effect of flushing on yearly maximum biomass predictions ... 199
16.4. Effect of dredging on yearly maximum biomass predictions ... 200
TABLES

S.1. Depth and Background Extinction Coefficients for Lakes in Eutrophication Study ........................................... xii
S.2. Effect of Complete Mixing on Yearly Maximum Biomass Predictions ............................................................ xix
3.1. Some Important Yearly Average Concentrations in Water and Bottom of Grote Rug, Ring 2, 1977 ....................... 21
5.1. Required Multiplication Factor of Molecular Diffusion Rates to Calibrate the Nutrient Model ......................... 52
5.2 Computed Annual Nitrogen and Phosphorus Fluxes in gr/m²-yr for the AL Dosed Ring 2 of Grote Rug in 1977 ....... 53
6.1 Monthly Average Values of Silicon in Rijn and IJsselmeer Water for the Period 1976-1978 ........................... 67
7.1. Names of Groups and Species in BLOOM II and Abbreviations Used in this Volume ..................................... 77
7.2 Ratio of R(T) to Pmax(T) at 20 deg C and Increase per 10 deg C Temperature Rise (Q10) of R(T) for Several Groups of Species .......................................................... 86
8.1. Species Coefficients to Calculate Pmax(T) and R(T) ......... 100
8.2. Stoichiometric Constants of Species in BLOOM II, Species Coefficients for Extinction, and Dry Weight to Chlorophyll Conversion ......................................................... 100
8.3. Yearly Constant Lake-Specific Inputs .......................... 105
9.1. Summarized Results for Limiting Factors and Species Composition as Observed and Predicted in the Nominal Runs of BLOOM II ......................................................... 110
11.1. Summary of Inputs for the Dissolved Oxygen Model .... 152
12.1. Parameter Values Established by Calibration .................. 157
13.1. Sources of Input Data for Lake Simulations .................. 160
13.2. Additional Comments to the Input Data for Lake Simulations .. 162
13.3. Summary of Oxygen Results Based upon Measured Chlorophyll Data ....................................................... 187
16.1. Effect of Complete Mixing on Yearly Maximum Biomass Predictions ......................................................... 206
16.2. Impacts of Promising Tactics on Oxygen, Based upon Biomass Computations by BLOOM II ................................. 210
Chapter 1

GENERAL INTRODUCTION

1.1. EUTROPHICATION IN THE NETHERLANDS

According to Parma [1.1], "Eutrophication is the process in water during which the factors stimulating autotrophic production become optimal." In contrast to many other definitions, the one by Parma (1) includes weeds in addition to phytoplankton, (2) considers more than one or two major nutrients as potential limitations, and (3) considers natural as well as human causes for eutrophication.

Because many factors are involved, the symptoms of eutrophication are highly site and time specific. Also, the magnitude of the problems depends on the function of a particular body of water: drinking water reservoirs, fish ponds, recreational lakes, and lakes in nature reserves have such different functions that what is acceptable or even desirable in one type of lake is objectionable in another.

According to Lund [1.2] and Vol. III, some common, objectionable symptoms of eutrophication are:

1. Bad appearance of the water (color: green "soup"; visibility: reduced to a few decimeters).
2. Bad odors (e.g., of sulfides), if a bloom dies off suddenly (a collapse).
3. High diurnal fluctuations in the concentration of dissolved oxygen when there is a bloom, because respiration of phytoplankton continues during the night, while primary production is confined to the daytime.
4. A major risk of anaerobic conditions after a collapse, possibly for several days, leading to massive killings of organisms such as fish and shellfish.
5. A low diversity of the ecosystem.
6. Clogging of filters at water intakes for industrial or drinking water purposes.
7. Toxic effects on other organisms, although literature information on this subject is rather scarce.

Eutrophication problems in the Netherlands are much more serious than in many other countries for several reasons:

- The country is (and in recent times has always been) densely populated: man's influence is extremely large.
- A relatively large fraction of the country is covered by water (rivers, canals, natural and man-made lakes, ditches, pits).
Most of these waters are very shallow (on the order of 1 to a few meters): no nutrients are removed to a hypolimnion, where they would become temporarily unavailable for growth of primary producers. Quite the opposite is true; as will be shown in Part I, occasionally Dutch bottoms are an excellent source for nutrients.

Concentrations of nutrients in many waters are to a large extent determined by the loadings from rivers, of which the most important (Rijn and Maas) happen to be heavily loaded with nutrients.

Pollution of these rivers is an international problem because they are border-crossing rivers. Hence there is a limit to the attainable nutrient reductions that may be achieved by Dutch management tactics. Further reductions are only possible in cooperation with other countries.

1.2. WABASIM PROJECT

1.2.1. Origins

In February 1953, extensive territories in the southwestern part of the Netherlands were inundated, when an exceptionally high tide coincided with a severe northwestern storm, lasting for several days. Especially the province of Zeeland suffered heavily and the number of casualties rose to over 1800. To improve the security of the country and prevent a repetition of the 1953 events, a Delta plan was proposed calling for the dikes to be rebuilt to higher levels than before the disaster, and for dams to close off most of the estuaries from the North Sea.

In the mid-1970s, public concern for the rich and rare ecology of the one remaining unclosed estuary (the Oosterschelde) forced reconsideration of the then partly completed plan. Naturally, the constructions already accomplished had dramatically changed the ecological conditions in many waters in the Delta region, and further construction could be expected to cause still more change. Accordingly, the Deltadienst (the government agency charged with overseeing the Delta plan) was instructed by Parliament to seek alternatives to damming the Oosterschelde that would preserve its ecological values.

In response to Parliament's directive, a Policy Analysis of the Oosterschelde (POLANO) study was carried out, during which a number of mathematical models of different aspects of the ecology were developed [1.3-1.6]. These models were used to estimate the probable ecological effects of the various proposed alternatives to the original Delta plan.

Following the completion of the POLANO study, and recognizing the desirability of optimizing the management of the aquatic ecosystems, the Deltadienst's Environmental Division initiated the Water BASIn Models (WABASIM) project in 1977, to build upon the models and
experience developed in POLANO. WABASIM is a multidisciplinary joint project of the Environmental Division and the Environmental Hydraulics Branch of the Delft Hydraulics Laboratory, with technical assistance from Rand.

1.2.2. Initial WABASIM Goal

The first goal of WABASIM is to develop a eutrophication model for the freshwater storage reservoir "Grote Rug" near the city of Dordrecht. This reservoir contains three butyl rubber enclosures (called Rings) with a diameter of 46 m each and the same water regime as the main reservoir, which receives its water from the Wantij, a branch of the Rijn. Since their installation in 1975, extensive weekly or even daily measurements have been made of biological and chemical variables including concentrations of nutrients, chlorophyll, wet weights of phytoplankton and zooplankton, solar intensities, extinction coefficients, and primary production ($^{14}$C method$^2$).

The Grote Rug project of the Delta Department and the Netherlands Institute for Drinking Water Supply (RID) was intended as a full-scale experiment to compare the results of the in-water removal of the nutrient phosphorus. Ring 1 and the main reservoir are dosed with $2^+$ Fe, Ring 2 is treated with $Al^{3+}$, and Ring 3 is a control, receiving water without any treatment. It was expected that phosphorus concentrations and hence phytoplankton levels would be lowest in Ring 2, because of the high affinity of Al for $PO_4^{3-}$; phosphorus concentrations should also be low in Ring 1, and Ring 3 of course was likely to have the highest phosphorus concentrations and hence largest phytoplankton blooms.

Indeed, in the years 1975 through 1978 the lowest concentrations of both orthophosphate and total phosphorus have been observed in Ring 2, followed by Ring 1, while the average concentrations in Ring 3 were much higher than in the other two rings. Unfortunately, the expected one-to-one relationship between phosphate and phytoplankton biomass apparently was incorrect, as blooms have occurred more frequently and with greater magnitude in Ring 2 than in any of the other reservoirs.

Notwithstanding the (unexpected) results, the measurement program was improved and extended over the years in order to obtain sufficient field data to generate and test hypotheses about the eutrophication process. Both quality and quantity make the present Grote Rug data bank particularly suitable for developing mathematical models.

Two different models are being developed to achieve this goal: one to model phytoplankton under given chemical (and other) conditions (BLOOM II) and one to model lake chemistry under given biological conditions (CHARON). An integrated version of the two models, which currently exists, was not available when the models were used for PAWN.
1.3. USE OF WABASIM MODELS IN PAWN

A regional survey of water quality problems [1.7] has indicated that eutrophication standards for nutrients and biomass\(^3\) are frequently violated in many Dutch waters. As the impacts tend to be worse in waters with high residence times (a few months or more), in this study, we shall consider only semistagnant waters: lakes and some (temporary) look-alikes (a stuwpand in the Maas; the Haringvliet).

There would be no concern about increasing nutrient concentrations, if these would not enhance phytoplankton blooms. Therefore modeling phytoplankton (by the WABASIM model BLOOM II) is central in our approach to designing eutrophication control strategies. This model computes size, time, composition, and limiting factors of (potential) blooms.

Some chemicals (nutrients) act as potential bloom-limiting constraints. If the chemical environment were invariant, or a simple function of time such as the amount of solar radiation or the water temperature, it would be easy to specify the abiotic conditions for now and for the future. Unfortunately this is not the case; water chemistry strongly depends on (1) the hydraulic conditions, and (2) the biological conditions. Therefore we would prefer to link our phytoplankton model to a model for water chemistry (such as the WABASIM model CHARON) and a network model for computing nutrient flows into lakes, such as the PAWN Distribution Model (Vol. XI) or Managerial Strategy Design Model (Vol. V).

One of the most important consequences of a phytoplankton bloom is its effect on the dissolved oxygen concentration. No WABASIM model considers this parameter, so as a part of PAWN we have developed a dissolved oxygen model called OXYMOD. The (potential) impacts of blooms on the oxygen concentrations in the water could be computed using information from BLOOM II as inputs to OXYMOD. All these interactions are shown in Fig. 1.1.

For lack of time, knowledge, and data, however, it was impossible to develop and integrate all of these models for PAWN. It was not possible to incorporate water quality processes sufficiently well into the network models to compute nutrient loadings to lakes under alternative conditions. There were insufficient data for most lakes to enable CHARON to compute the important nutrient exchange rates between the water and bottom, although it could be shown how important these transports are. Also an integrated version of CHARON and BLOOM II to compute the important feedbacks between chemical and biological processes could not be finished in time. A calibrated version of BLOOM II was, however, available, and computation of the oxygen budgets of the lakes was made possible by using the newly developed model OXYMOD, which receives some of its inputs from BLOOM II.

The methodology of BLOOM II, including most of the references, is published by Los [1.8] as part of the WABASIM project and will only
Fig. 1.1--Relations among eutrophication methodology components (inputs to these models from sources other than these models are not shown)
be summarized here. The methodology of the PAWN model OXYMOD and the WABASIM model CHARON is first published here, although briefly for the latter.

1.4. SELECTION OF PAWN LAKES

For an overview of eutrophication in the Netherlands, it is important to select lakes with different characteristics in different parts of the country. Also a sufficient amount of (high quality) data must be available. Based upon these criteria, the following lakes have been considered in the PAWN eutrophication study. Their locations can be seen in Fig. 1.2.

IJsselmeer
Markermeer
Veluwemeer
Wolderwijd
Gooimeer
IJmeer
Slotermeer
Westeinderplassen
Haringvliet
Stuwpond Lith
Zoommeer

1.5. PHYTOPLANKTON

1.5.1. Requirement for Solar Energy

All plants require raw materials (nutrients) and energy in order to grow. The sun provides the energy, at a rate per square meter of surface area that depends upon latitude, cloud cover, time of day, and season of year. The energy must be shared among all the phytoplankton floating in the water column below that square meter of surface area, with an allowance set aside for reflection from the water surface and absorption by the bulk water and its contents other than phytoplankton. The more phytoplankton there are, the less solar energy is available for each, until the energy per phytoplankton cell is too small to sustain growth. At that point, solar energy becomes a growth-limiting factor.

1.5.2. Nutrient Requirements

Plants also require about a dozen chemical elements for a normal development, among which are nitrogen, phosphorus, sulfur, calcium, potassium, magnesium, and iron. The requirements for each element vary widely and, particularly in terrestrial ecosystems, elements which are only required in small amounts (trace elements) are frequently limiting. Because aquatic systems are much more homogeneous, and usually receive water from sources which are heavily loaded with many chemical elements, it is rather unlikely that trace elements will limit phytoplankton in eutrophic waters, although the possibility cannot be ruled out completely. Thus, we have not included trace elements as potential phytoplankton biomass limiting factors.
Fig. 1.2—Lakes investigated in PAWN's eutrophication study
Four macronutrients are often reported as limiting factors: carbon, nitrogen, phosphorus, and silicon, of which the last is essential only for a single group of phytoplankton species, diatoms. These species use silicon to build strong skeletons surrounding the cell walls. As calculation with the nutrient model CHARON (see Part I below) has shown that depletion of carbon is rather unlikely under typical Dutch conditions, it is not included as a limiting factor in our study. This leaves the three major nutrients, phosphorus, nitrogen, and silicon, which along with solar energy are considered in the study to be potential growth-limiting factors for phytoplankton.

Nitrogen and phosphorus are vital to all phytoplankton species. Nitrogen is an essential component of cell proteins such as enzymes, for genetic material, and of light-sensitive pigments like chlorophyll-a, which are used for fixation of solar energy. Because of its importance to many vital physiological processes, nitrogen deficiencies cannot be tolerated for long.

Phosphorus is an important component of proteins, nucleic acids, and lipids (e.g., in the cell walls). Coupling and uncoupling of orthophosphate groups to certain sugars are the main reactions by which chemical energy is stored or released in the cell. Phytoplankton are usually less sensitive to phosphorus than to nitrogen deficiencies, hence survival at extremely low internal phosphorus concentrations is often possible for some time.

Nutrients appear in many forms in a phytoplankter's environment, of which usually only a fraction can be assimilated. An even smaller number of forms may be used directly in metabolic processes. But this does not imply that any nutrient is limiting as soon as the metabolically preferred forms are depleted. Chemical and bacterial processes in the water may convert one form of a nutrient into another, phytoplankton species may sometimes shift their uptake preference between different forms, and phytoplankton can often internally convert one form of a nutrient into another by enzyme-catalyzed reactions. For example, NH$_4^+$ is often preferred as a nitrogen source, but many species can internally convert NO$_3^-$, NO$_2^-$, or various organic nitrogen compounds to NH$_4^+$, and some species of blue-green phytoplankton can even use atmospheric N$_2$.

Those fractions of a nutrient which may be used directly or after some rapid conversion process will be called available, in contrast to the unavailable fractions, which either cannot be used or can only be used after some slow conversion process.
1.5.3. Maximum Potential Phytoplankton Bloom

We have constructed a model (BLOOM II) based on the technique of linear programming, which calculates the maximum potential phytoplankton bloom consistent with the prevailing abiotic conditions. This model, which is described more fully in Part II of this volume, finds the combination of phytoplankton species which can most effectively use the available solar energy and the available quantities of phosphorus, nitrogen, and silicon, to produce the largest possible biomass (measured as dry weight). Because factors not included in the model may sometimes be limiting, or because the growth history of the phytoplankton has resulted in a different combination of species being present than that predicted by the model, the predictions will tend to overstate the risk of large blooms. However, underprediction is also possible due to uncertainties in some model parameters, notably the mortality rate of phytoplankton.

BLOOM II has been used in the PAWN study to help estimate the effects of various possible eutrophication control tactics on the potential bloom maxima in the lakes selected for study (Sec. 1.4). But to use BLOOM II in this way, it was first necessary to estimate the effects of the control tactics on the available nutrients and solar energy. This is not so difficult for solar energy, but it proved to be a monumental task for nutrients, a task which we discuss in the following section.

1.6. NUTRIENT CYCLES

There is a continuing turnover of nutrients in the aquatic system. They may begin, for example, in dissolved form, and be incorporated into living phytoplankton. Upon dying, the phytoplankton release 50 percent of their nutrient content back to the dissolved form, and retain the remaining 50 percent in the suspended fragments of dead phytoplankton. (The 50-percent split is an assumption made in the model; see Chap. 7 and Ref. 1.8 for discussion.) The fragments of dead phytoplankton settle to the bottom, carrying their nutrients with them. Whether still suspended or on the bottom, these fragments can be mineralized (i.e., digested by bacterial action), a process that releases their nutrients into the water in dissolved form. Finally, chemical reactions may occur to bind nutrients to inorganic particles, either suspended or on the bottom. In this complicated system, both the concentrations of the various forms and their turnover rates may vary considerably with the kind of lake, temperature, dominant phytoplankton species, and many other variables. A limited number of these processes is schematically represented in Fig. 1.3.

No other organisms but phytoplankton and zooplankton have been included in the nutrient cycles, although occasionally fish or bottom organisms feeding on phytoplankton could be of importance. This is not because they ever contain a significant amount of nutrients, but because they may increase some of the turnover rates. Generally, however, the
Fig. 1.3 - Basic nutrient cycle used for phytoplankton and nutrient models.
influence of these organisms on the nutrient cycles in eutrophic lakes seems negligible.

Usually the flows of the nutrient cycles are described by differential equations, which, depending on the form of the equations, are solved analytically or numerically. Examples of this classical approach can be found in Di Toro et al. [1.9], Nyholm [1.10], Jorgensen et al. [1.11], and Bierman [1.12].

Many of these models were, however, developed for lakes with a depth of more than 10 m, which become stratified in summer and show little or no exchange between the hypolimnion and overlying water during the growing season. The only important nutrient transfer in such lakes is the sedimentation of living phytoplankton and detritus to the bottom; hence the rate of change of the total amount of nutrients in the water phase is small relative to the changes in phytoplankton kinetics. A simple approach, using first order equations has often been applied successfully, for instance, in modeling phytoplankton populations of the Great Lakes in the United States.

By contrast, many Dutch lakes are shallow. The average depth in some of the most eutrophic is less than 2 m. Their nutrient concentrations are often ten times higher than those in lakes to which the classical models were applied, and vary greatly sometimes even within short time intervals such as one week. Several Scandinavian authors (e.g., Nyholm [1.10] and Jorgensen et al. [1.11]) have used the classical approach to model similar lakes by empirically determined rate constants for the exchange transports between water and bottom, but the predictive value of this approach is questionable, since the parameters of the models seem specific to a particular lake and to a particular interval of time (i.e., calibrating the model to one lake and applying it to another, or calibrating it to data from one year and validating against data from another year, often yield very poor results).

It is an important conclusion of WABASIM that over the years observed nutrient concentrations in various Dutch lakes cannot be described by simple differential equation models with one set of rate coefficients. A much better result has already been obtained using CHARON, provided that the changes in biological phenomena are explicitly formulated as boundary conditions to this model and that no "explosive" bottom fluxes occur.

1.7. DISSOLVED OXYGEN

Perhaps the most objectionable consequence of a large phytoplankton bloom is oxygen depletion. The oxygen is partially depleted every night, since phytoplankton use oxygen for respiration throughout the entire 24 hours of the day but only photosynthesize oxygen during the daylight hours, when the necessary solar energy is available. This gives rise to the characteristic pattern of oversaturation of oxygen during the day, and undersaturation during the night—the so-called diurnal oxygen sag.
The degree of undersaturation achieved by the diurnal oxygen sag, however, is rarely a problem. Rather, the problem occurs when a large bloom suddenly dies off and the dead phytoplankton sink to the bottom. If such a collapse occurs at a time when temperatures are high enough to promote rapid mineralization (which consumes oxygen), and when the air is still enough to retard the movement of oxygen from the air to the water (re-aeration), then the water can be entirely depleted of oxygen. Oxygen-free conditions (called anaerobic) can result in massive fish kills. Furthermore, mineralization under anaerobic conditions results in toxic and ill-smelling compounds.

Not every large bloom collapses, and of those that do, not all collapse under the circumstances that favor anaerobic conditions. Thus, large phytoplankton blooms merely pose the risk of anaerobic conditions, and not the certainty.

To assess this risk, we constructed a model of the dissolved oxygen budget of a lake, called OXYMOD. The model is formulated in terms of differential equations, and calculates both the diurnal oxygen sag and the minimum oxygen concentration that would be achieved, week by week, if the bloom predicted by BLOOM II for that week were to collapse suddenly. Discussion of OXYMOD can be found in Part III of this volume.

1.8. ORGANIZATION OF THIS VOLUME

The remainder of this volume is divided into four parts. Part I discusses CHARON, the nutrient model being developed under the WABASIM project. Among the subjects discussed are model formulation, nutrient transport and exchange processes, calibration and validation, and some results.

Part II deals with BLOOM II, the phytoplankton model that, like CHARON, is being developed under the WABASIM project. It outlines the formulation of the model, data requirements, calibration and validation. Results and conclusions are mostly deferred to Part IV.

Part III treats OXYMOD, the dissolved oxygen model developed under PAWN. Subjects dealt with include formulation, data needs, calibration, and recommendations for further development.

Part IV presents results and conclusions.

NOTES

1. Phytoplankton (and plants in general) are called autotrophic as opposed to heterotrophic animals, because they obtain energy for maintenance and reproduction directly from the sun, rather than
by eating other organisms.

2. Organic carbons are used for storage of energy, which is fixed from the sun by photochemical reactions in the plant's cell. Measuring uptake of labeled carbon is one way to measure the rate of energy fixation.

3. For the new "Indikatief Meerjaren Plan" (IMP), RIZA suggests using a provisional standard concentration of 100 mg chlorophyll/m$^3$ and a target concentration of 50 mg chlorophyll/m$^3$. Chlorophyll is the only commonly measured phytoplankton biomass indicator. These standards will be discussed briefly in Sec. 16.2.

4. Trace elements are included in CHARON insofar as they are important to the water chemistry. They can easily be included in both CHARON and BLOOM II if they appear to be important to phytoplankton blooms.

5. "Rapid" conversion means that a substantial part of an unavailable form can become available during one time-step.

6. An unusually large "explosive" release of nutrients from the bottom; this process will be described in more detail in Sec. 3.1.2.

REFERENCES


1.10. Nyholm, N., "A Simulation Model for Phytoplankton Growth and


PART I: THE NUTRIENT MODEL (CHARON)
Chapter 2
INTRODUCTION

One of the models being developed under the WABASIM project (Sec. 1.2) is the nutrient model CHARON, an extension of Rand's chemical equilibrium model. We felt it necessary to have a nutrient model because reduction of phosphorus loadings is the primary tactic currently being considered in the Netherlands to control eutrophication. We felt it necessary to develop our own model of nutrient processes, rather than to apply a previously existing model, because the situation in Dutch lakes is highly unusual. Previous models have been designed to apply to relatively deep lakes which become thermally stratified during the summer. The hypolimnion then acts as a nutrient trap, which can be modeled easily and simply. By contrast, many Dutch lakes, including the most eutrophic, are too shallow to become stratified. While nutrients, especially phosphorus, are indeed sequestered in the bottom sediments of these lakes, they can be released in significant, even massive, quantities under the proper conditions.

Because chemical reduction of bottom sediments is a particularly important trigger for nutrient release, we thought that an oxidator balance model could be used to indicate where and when to expect this nutrient release to the water system. The model could even be extended to a complete nutrient model. However, there were insufficient data describing bottom sediment compositions and oxidator fluxes to the bottom for the application of this model to the PAWN lakes.

We applied the nutrient model to the well-documented Grote Rug system (Sec. 1.2) and formulated some conclusions and hypotheses, which may also hold for other shallow Dutch lakes with comparable nutrient loadings. These computations also indicate the symptoms that accompany a phosphorus release from the bottom. Indeed, the symptoms accompanying high release rates of phosphorus were detected in some of the PAWN lakes.

Moreover, it became obvious that the entire chemical system of the water is necessary to describe the behavior of nutrients. Unfortunately, this implied that infrequently measured variables such as Ca++ and alkalinity are very important.

In Part I of this volume we shall discuss:

- Processes involving the nutrients carbon, nitrogen, phosphorus, and silicon, and pH, including the exchange of
chemical species between the bottom and the overlying water.

- Model formulation with special emphasis on the extensions to Rand's original chemical equilibrium model CHEEQ.
- Calibration and validation to Grote Rug.
Chapter 3

NUTRIENT PROCESSES

Nutrient modeling in PAWN is not done for its own sake, but for the light it can shed on the eutrophication problem. How paramount nutrient availability is to the prediction of phytoplankton biomass in a water column will be thoroughly discussed in Part II on phytoplankton modeling. Nutrient availability is determined by:

- Loadings to the system.
- Exchange reactions of the water with the atmosphere and the bottom.

Exchange with the atmosphere is possible for only one nutrient, nitrogen. There can be a considerable exchange of all nutrients, however, between the bottom and water phase. Modeling the nutrient phosphorus will be the most important subject in the next sections.

Assuming that loadings to the water are known, the main problem is to describe the exchange reactions between water and bottom. Contrary to the picture suggested by the common expression "decay of phosphorus," this element is in fact conservative if we include the bottom in our considerations. Phosphorus concentrations in natural bodies of water diminish by sedimentation of organic and inorganic phosphorus. Organic sedimentation is mainly from living and dead phytoplankton (detritus), and hence this process is directly related to phytoplankton blooms.

Phosphorus sediments mainly in the form of suspended materials to which phosphorus is adsorbed, or with which it precipitates. The amounts of precipitates and adsorbed materials depend on the chemistry of the water system. Phosphorus can return from bottom sediments to the water by resuspension of suspended materials, and by dispersion of dissolved material. In this chapter we shall show how the availability of phosphorus and other nutrients is influenced by the characteristics of the bottom and by chemical reactions in the overlying water.

3.1. EXCHANGE REACTIONS WITH THE BOTTOM

The exchange rate of materials between bottom and overlying water is one of the greatest existing uncertainties in nutrient modeling. Especially for phosphorus there is no consensus about the nature or the rate of the exchange reactions. Theis and McCabe [3.1] report a range of literature values for the release of phosphorus from 0.03 to 30 mg P m$^{-2}$ day$^{-1}$. All that is well established is that the release rates for anoxic sediments are higher than for oxic sediments.
Mass balances have been determined for many lakes in the Netherlands, e.g., for the eutrophication study of the Randmeren [3.2]. These mass balances show the net result of exchange reactions with the bottom. Frequently sedimentation exceeds fluxes from the bottom, but periodically releases from the bottom are higher. Since these mass balances give only the net results, they give no information about the absolute amounts of sedimentation or release.

Phosphorus is an important element in many chemical and biological processes, which have been studied extensively. Chemical investigations, which include those by Nriagu [3.3], Lyklema [3.4], Hietjes [3.5], and several others, clearly show that sorption of phosphorus to oxides, hydroxides, and solid Ca phases are particularly important mechanisms. There is uncertainty as to the kind of chemical or physical mechanisms which are involved in these processes. In general, however, sorption seems more important in oxic environments, but under anoxic conditions formation of iron-phosphates is responsible for the observed behavior of phosphorus.

For the PAWN project we have tried to calibrate the fluxes to and from the bottom with measured concentration data for the overlying water and for the interstitial pore waters. This calibration is possible only if the intensity of both processes is not the same throughout the year, which is indeed the case according to the mass balance studies for several lakes. These investigations have demonstrated that we may distinguish two different kinds of nutrient fluxes from the bottom:

- A (temperature dependent) normal flux.
- An explosive flux from the bottom which abruptly releases large amounts of nutrients.

3.1.1. Normal Fluxes

For most chemical species which are dissolved in the water, there is a concentration gradient between the overlying and interstitial pore water. Available interstitial pore water data for Grote Rug Ring 2 in 1977 [3.6] gave us an indication of the order of magnitude of the existing gradients (Table 3.1). Clearly there is a gradient favoring a movement of orthophosphate ($PO_4^{3-}$) out of the bottom into the overlying water, and it is a truism of thermodynamics that where there is a gradient there must be a flux. In most lakes, including Dutch lakes, a flux due to sedimentation counterbalances this flux so that the average net movement of $PO_4^{3-}$ is from water to bottom. But if the counterbalancing flux stops or slows, even temporarily, then one should expect the bottom to serve as a net source of $PO_4^{3-}$ for phytoplankton.
Table 3.1
SOME IMPORTANT YEARLY AVERAGE CONCENTRATIONS IN WATER AND BOTTOM OF GROTE RUG, RING 2, 1977

<table>
<thead>
<tr>
<th>Species</th>
<th>Unit</th>
<th>Overlying Water</th>
<th>Oxidized Layer Bottom</th>
<th>Reduced Layer Bottom</th>
</tr>
</thead>
<tbody>
<tr>
<td>PO₄³⁻</td>
<td>(ppm)</td>
<td>0.003</td>
<td>0.1048</td>
<td>0.1832</td>
</tr>
<tr>
<td>H₄SiO₄</td>
<td>(ppm)</td>
<td>0.5</td>
<td>2.57</td>
<td>5.14</td>
</tr>
<tr>
<td>NH₃</td>
<td>(ppm)</td>
<td>0.1</td>
<td>3.09</td>
<td>4.42</td>
</tr>
<tr>
<td>NO₃⁻</td>
<td>(ppm)</td>
<td>2.22</td>
<td>0.01</td>
<td>0.00</td>
</tr>
<tr>
<td>alk</td>
<td>(meq/l)</td>
<td>2.0</td>
<td>3.54</td>
<td>5.17</td>
</tr>
<tr>
<td>CO₂</td>
<td>(satur%)</td>
<td>10%</td>
<td>2000%</td>
<td>3000%</td>
</tr>
</tbody>
</table>

NOTE: The mean depth of the oxidized layer is about 2.0 cm; alk = alkalinity.

Some of the counterbalancing flux is due to sedimentation of inorganic particulate matter on which phosphorus is adsorbed, and some is due to the sinking of dead phytoplankton to the bottom. During the rapid growth phase of a bloom, both components of sedimentation would be reduced, since few phytoplankton would be dying and the growing phytoplankton would deplete the water of phosphorus, leaving less to be adsorbed on inorganic particles. In sum, the occurrence of a phytoplankton bloom should promote the conditions necessary for a net phosphorus flux out of the bottom.

Measures to reduce the amount of phosphorus in the overlying water will also tend to increase the net flux from the bottom, since less phosphorus in the water means that less can sediment. In addition, lowering the phosphorus concentration in the water will increase the gradient from bottom to water, and hence increase the outward flux. That is, measures to reduce the phosphorus content of lakes are to some extent self-defeating.

But to what extent? While we may be sure that a flux of phosphorus occurs from the bottom, we cannot be sure of its rate. In a later chapter, when we discuss the calibration of CHARON to Grote Rug data, we will conclude that the rate is substantial.

3.1.2. Explosive Fluxes

3.1.2.1. Chemical Factors. Mortimer [3.7] has shown that chemically reduced bottoms release considerably larger amounts of phosphorus than do oxidized bottoms. His explanation is that solid Fe(OH)₃ is reduced and that phosphates which are adsorbed to this iron compound dissolve under reduced conditions. In the Netherlands
this theory was not accepted as an explanation for high fluxes from the bottom, since it was considered highly improbable that the bottom of a shallow lake would become anoxic. It was commonly believed that high values of the pH of surface waters were responsible for high P fluxes, a theory based on correlation (e.g., Golterman and Wisselo [3.8]). Indeed Mortimer developed his theory for the bottoms of stratified lakes with an oxygenless hypolimnion. However, we believe that Mortimer's theory can be applied to the bottoms of shallow Dutch lakes.

The bottom contains a certain amount of organic material, mainly from phytoplankton, which decays at a rate more or less independent of what is going on in the surface water. This decay reaction produces electrons which are accepted by any available electron acceptor according to the following preferences: first $O_2$, then $NO_3^-$, $SO_4^{2-}$, and finally $CO_2$ and $H_2O$, which are converted to methane ($CH_4$). Little is known about the rates of these reactions.

According to Di Toro [3.9], the rate of decay is independent of the available electron acceptor, but he mainly studied nitrate, sulfate and $CO_2$ oxidation rather than $O_2$ oxidation processes. According to Cappenberg [3.10], the rate of $O_2$ oxidation must be orders of magnitude higher than the rate of nitrate oxidation.

It is, however, clear that there must be a constant flux of electron acceptors (also called oxidators) from the surface water to the bottom. Some indication of the magnitude of this flux can be obtained from data on oxygen consumption by bottoms. Under steady state conditions the flux of electron acceptors to the bottom can be determined from the flux of organic material to the bottom, for as mentioned above, the decay of organic matter produces the electrons for which acceptors are demanded. Whatever kind of transport processes exist (molecular diffusion, mixing, or bioturbation), the magnitude of these fluxes depends on the concentration gradient over the exchange layers.

We have some measurements of interstitial water and overlying water concentrations from Grote Rug. For the following reasons, these give only an indication of the state of the bottom and the processes occurring there.

- Interstitial water measurements fluctuate strongly, probably because the bottom is not homogeneous over the horizontal area, and because it is impossible to sample the same spot more than once.
- The thickness of layers varies strongly, or is not even measured.
Only two layers (oxidized and reduced) are distinguished, and
the depth of the sample of the reduced layer is not indicated
at all.

Gradients within layers are not measured, and concentration
measurements are only mean values.

Nevertheless, some conclusions can be drawn. Oxygen is always
undetectable, even in the oxidized layer, and nitrate and nitrite
were detected only in one ring, during winter. Hence there must be a
considerable gradient for both oxidators.

The chemical conditions in an environment in which nitrate is the
oxidator differ from those in an oxygen or sulfate environment.
Measurements by Berner [3.11] show a distinct order in which
electron acceptors take up the electrons: \( O_2 \) is preferred to \( NO_3^- \)
is preferred to \( SO_4^{2-} \) is preferred to \( CO_2 + H_2O \). There is a drop
in redox (short for reduction/oxidation) potential whenever one
oxidator is exhausted and the next must be employed.

One oxidator has not been mentioned: \( Fe^{3+} \). In bottom sediments
iron is mainly present in solid forms. In oxidized environments it
is present as three valent \( Fe(OH)_3 \), but in reduced environments as
two valent \( FeS \) and \( FeCO_3 \). The solubility of two valent solid iron
phases is usually much higher than of three valent iron phases.
Consequently the total solubility of iron is higher in a reduced than
in an oxidized environment.

The ratio of \( Fe^{3+} \) to \( Fe^{2+} \) in a solution depends on the redox
potential. In surface waters this ratio is normally greater than
1.0. In interstitial water this ratio is smaller than 1.0. But even
in an anaerobic environment in the presence of nitrate, the amount of
\( Fe^{2+} \) in solution is low, although orders of magnitude higher than
under aerobic circumstances. The reason is that the concentration of
\( Fe^{3+} \), which is determined by the solubility of \( Fe(OH)_3 \), is
extremely low. A considerable concentration of \( Fe^{2+} \) and total iron
can be found only if the \( SO_4^{2-}/S^- \) couple determines the redox
potential, since under this condition \( Fe^{3+} \) will be reduced to
\( Fe^{2+} \), and \( Fe(OH)_3 \) will dissolve. But this happens only after the
preferred oxidators, \( O_2 \) and \( NO_3^- \), have been exhausted.
Schippel [3.12] points out that high exchange rates of phosphorus coincide with the formation of \( S^{2-} \), which confirms the idea that \( \text{Fe(OH)}_3 \) can only dissolve and liberate adsorbed P when the redox potential (symbol \( \text{Eh} \)) is low enough to reduce sulfate. However, as we show next, physical as well as chemical properties are important in determining exchange behavior of bottoms.

3.1.2.2. Physical Factors. In soil sciences, iron hydroxides have always played an important role in studying soil genesis and classification, as shown for example in [3.13] and [3.14]. The solubility of Fe is mainly determined by pH and \( \text{Eh} \), which both vary vertically in soils. Thus dissolution, transport, and precipitation of iron oxides and hydroxides are among the main processes which are studied in soil sciences.

Freshly formed precipitates of iron hydroxides are usually not crystalline but amorphous and form layers around and between solid particles such as feldspars, quartz grains, etc. Thus podsolic soils, which are formed in the Netherlands on Pleistocene sand deposits as well as in other countries, show a brown layer 5 to 20 cm thick at a depth of about 30 to 60 cm called iron pan, or in Dutch, oorlaag.

Often it is very hard to penetrate these layers, which consist of cemented quartz grains. Although the color of the sediments indicates the presence of \( \text{Fe(OH)}_3 \), usually less than 0.5 percent of the total weight consists of it. Dissolution of these minor amounts of amorphous material causes a disaggregation of particles and alters the physical properties of solid materials.

An indication of how easily such transformations can be brought about can be calculated as follows. We shall compute the oxygen equivalents necessary to reduce and dissolve 0.5 weight percentage of \( \text{Fe(OH)}_3 \) from a 1 m², 1 cm thick bottom layer. Assuming a sediment density of 1.5 and that 30 percent of the sediment consists of solid material, the total amount of solid sediment is 3.5 kg or 22.5 g of \( \text{Fe(OH)}_3 \). To reduce this amount of iron (0.21 mole), we need 0.053 mole or 1.68 g oxygen equivalents per m². Since the total oxidizer flux from the overlying water to the bottom is on the order of 1.0 to 2.0 g oxygen equivalents per m² per day, only small changes in the fluxes are sufficient to reduce and dissolve the \( \text{Fe(OH)}_3 \) coatings. If the oxidator flux is less than the decay of organic material (both in oxygen equivalents), Fe solubility greatly increases.

A larger amount of oxygen equivalents is, however, necessary to reduce all the iron in bottom sediment, because an iron content of 15.0 percent is not uncommon. Not all of the iron occurs as coatings, though. A decrease in iron can have a considerable impact on the chemistry of the system, e.g., enlarging the solubility of \( \text{PO}_4^{3-} \).

But the physical characteristics may be changed as well. Sufficient
amounts of Fe(OH)$_3$ to change the physical properties of the system, however, can only dissolve when oxygen or nitrate are not available as oxidators. Whether iron hydroxides indeed dissolve depends on the availability of electron acceptors, hence:

- The flux of oxygen and nitrate to the pore water.
- The decay rate of organic material.

The first process depends on the oxygen and nitrate concentration in the bulk water and on the dispersion and hence temperature. The second process depends on temperature and probably on the amount of electron acceptor. The latter could imply that decay by nitrate oxidation is much slower than by O$_2$ oxidation. Still it is conceivable that the oxidator is used more rapidly than supplied by transport of oxygen and nitrate. In that case Fe$^{3+}$ and sulfate will be used as the oxidator; hence the oxidized layer becomes smaller and smaller. When no oxidized layer exists, or the layer is very thin, the overlying water comes into contact with reduced, noncemented sediments.

In his discussion on shallow eutrophic lakes in Florida, where no defined sediment-water interface exists, Brezonik [3.15] describes this situation as: "A flocculent suspension (covers the bottom in these lakes) with gradual compaction from thin soup to consolidated sediment, occurring over a depth of perhaps several feet." He also states that: "The role of the oxidized microzone in influencing sediment-water nutrient cycling would be a fruitful area for further research." Ripl [3.16] performed experiments in which he dosed lakes with nitrate. His results suggest that a higher availability of nitrate suppresses the P fluxes from the bottom. Because oxidation of O$_2$ proceeds much faster than oxidation by nitrate, it cannot keep the oxidized layer thick enough.

These differences in physical properties between reduced and oxidized bottom sediments are also observed in Dutch lakes [3.17]. Under reduced conditions there is probably more mixing. Thus the magnitude of many processes changes considerably, and sometimes the processes even change direction. The kind of exchange reaction promoted by the oxidized layer becoming reduced is here defined as an "explosion." Although the flux of oxidator from overlying water to the pore water will also be increased, it will take at least some time before the bottom sediment is restored to a well-consolidated oxidized layer.

### 3.2. NUTRIENT REACTIONS IN WATER

In this section we shall treat the important reactions in water which influence nutrient behavior. Because the value of the pH is mainly determined by reactions of carbon and phosphorus behavior is strongly determined by pH, carbon must also be modeled, although it is not used as a nutrient in our phytoplankton model.
3.2.1. Carbon

The pH, which is the main variable for the chemical equilibrium system, is determined by the C fluxes in the system, unless a bottom explosion is taking place. The driving forces in the carbon system are the uptake and release of CO$_2$ by phytoplankton. Unfortunately phytoplankton biomass is usually measured as mg chlorophyll per m$^3$, which forces us to use a conversion between carbon and chlorophyll to compute carbon fluxes. These problems are discussed in Sec. 8.2.1. Inaccuracies in this conversion can have a substantial impact on pH (e.g., it is obvious from oxygen and pH measurements in the undosed Ring in 1977 that the carbon fluxes were high at the middle of the year, although this is not obvious from the measured chlorophyll levels). Carbon fluxes related to phytoplankton dynamics are an input to the chemical model. Only the rate constants had to be specified for the carbon fluxes (1) between water and air and (2) from the bottom.

3.2.1.1. CO$_2$ Transfer between Air and Water. CO$_2$ transport between air and water is described by the following equation:

$$\text{Flux} = \text{DELT}t \times K \times (\text{xmf(CO}_2) - \text{xmf(sat)}),$$

(3.1)

where xmf denotes concentration expressed as mole fraction. A plus sign for the flux implies an increase in the slow reactant, which in this case is the CO$_2$ in the air. We have computed K as:

$$K = k \times \text{wind}^2 \times 1.016 \times T^{-20}$$

where wind = wind speed at 10 m above water level in m/s, 
k = transport coefficient, 
T = temperature (°C).

This equation was developed for oxygen in [3.18], but no indication was found in the literature that it could not also be used for CO$_2$. The main problem is to obtain the value of the transport coefficient, which for any gas depends on the molecular diffusion coefficient and on eventual reactions in the diffusion layer. In the case of CO$_2$, these reactions indeed occur:

$$\text{CO}_2 + \text{OH}^- \rightarrow \text{HCO}_3^-$$

(3.2)
\[ CO_2 + H_2O \rightarrow H_2CO_3 \]  \hspace{1cm} (3.3)

Between pH 7.0 and 9.5 HCO$_3^-$ is the most important carbon species in the equilibrium system. According to [3.19], the rate constant for reaction (3.3) is 8500 liter/mol·sec, which does not seem high enough to change the gradient in the diffusion layer considerably at pH=8, but could possibly double the gradient at pH=9. Several authors, e.g., Ref. 3.20, suggest that the rate constant as it is measured under ideal, chemically clean conditions, is a lower bound and that it will be higher in the presence of catalysts such as phosphate, borate, or carbon anhydrase.

CO$_2$ and O$_2$ fluxes differ considerably. According to the equation for gas exchange, the rate of a flux is equivalent to the difference between the equilibrium and the actual concentration. With an equilibrium concentration of $10^{-5}$ moles/l for CO$_2$, the maximum flux rate into the water at complete undersaturation is equivalent to $10^{-5} - 0.0$. But the flux rate in the opposite direction can be as high as $19\times10^{-5}$ at 2000-percent oversaturation. The equilibrium concentration of O$_2$, which is $3.12\times10^{-4}$ moles/l, is 31 times higher than for CO$_2$. Thus assuming the same transport coefficients for O$_2$ and CO$_2$, a 1.0-mole change in biomass at equilibrium, which means an uptake of 1.0 mole of CO$_2$ and the liberation of about 1.0 mole of O$_2$, would lead to a 31 times higher flux rate of O$_2$ than of CO$_2$. Thus O$_2$ will usually approach its equilibrium much more rapidly than CO$_2$, and a high over- or undersaturation of CO$_2$ is much more likely to occur. In the IJsselmeer, for example, pH values above 9.5 are not uncommon, implying a saturation level for CO$_2$ of only 5 percent. This could mean that the CO$_2$ concentration would be exhausted very quickly. However, as soon as CO$_2$ is removed by phytoplankton, the following reaction occurs:

\[ HCO_3^- \rightarrow OH^- + CO_2 \]  \hspace{1cm} (3.4)

and the pH starts rising, because:
\[(\text{H}^+)(\text{HCO}_3^-)/(\text{CO}_2) = \text{Constant}\] (3.5)

The change in the \text{H}^+ concentration is large in comparison to the change in \text{HCO}_3^-, unless the alkalinity is low. Therefore \text{CO}_2 will still be undersaturated, leading to a flux from the air. In other words, a high alkalinity is a major carbon source for phytoplankton, but a substantial flux of \text{CO}_2 from the atmosphere is not inhibited.

As an illustration, the alkalinity of about 2.5 meq of Rijn water is enough to support a biomass of about 900 mg of chlorophyll per liter.

A major side effect of a rise in \text{pH} caused by \text{CO}_2 uptake by phytoplankton is a decrease in the adsorption of phosphorus to suspended material. Hence a developing bloom can increase the availability of this nutrient significantly.

3.2.1.2. Carbon Fluxes from the Bottom. It is very difficult to estimate carbon fluxes from the bottom. It is also necessary to distinguish \text{CO}_2 from \text{HCO}_3^- fluxes. To maintain electroneutrality in the system, the \text{HCO}_3^- flux has to be compensated in terms of charges by a flux of cations, most likely of \text{Ca}^{2+}. However, except in summer at a high \text{pH} there is hardly any gradient between bulk water and interstitial water for \text{Ca}^{2+}. In summer the bulk water is saturated with respect to \text{CaCO}_3.

Hence we have not assumed a substantial \text{HCO}_3^- flux from the bottom. The impacts of bottom fluxes on \text{pH} are usually marginal since the overlying water is so well buffered with \text{HCO}_3^- . The exchange of \text{CO}_2 with the air is an order of magnitude higher. A few remarks can be added:

- \text{NH}_3 fluxes from the bottom result in nitrification of the bulk water and production of a strong acid, or destruction of alkalinity:

  \[\text{NH}_3 + 2 \text{O}_2 \rightarrow \text{H}^+ + \text{NO}_3^- + \text{H}_2\text{O}\]

  Uptake of \text{NO}_3^- by phytoplankton has the reverse effect.
Fluxes of reduced products such as $\text{H}_2\text{S}$ or $\text{CH}_4$ all have the same effect:

$$
\text{H}_2\text{S} + 2\text{O}_2 \rightarrow 2\text{H}^+ + \text{SO}_4
$$

$$
\text{CH}_4 + 3\text{O}_2 \rightarrow \text{CO}_2 + 2\text{H}_2\text{O}
$$

3.2.1.3. Carbon Loadings and Outflow. In our calibration and use of CHARON, we have computed the CO$_2$ concentration of the inlet water, which is not measured, from measurements of pH and alkalinity. The pH of incoming water is usually rather low, about 7.6.

If the water is oversaturated, CO$_2$ disappears into the air. But the amount which is exchanged with the air as a result of oversaturation of the inlet water is very small compared to the fluxes which are caused by the waxing and waning of phytoplankton. The oversaturation of the inlet water is important, however, since it decreases pH. The equilibrium composition of the inlet and lake water differs considerably. Without any phytoplankton the pH of the lake would be about 8.2.

3.2.1.4. Carbon Fluxes from BOD Loadings. BOD measurements are determined by two distinctive constituents:

- External loadings to the system of (dead) organic material.
- Internal loadings from phytoplankton growth in the system.

Both loadings are separated in the model. BOD loadings simply decay, a process by which CO$_2$ is produced and the pH decreased.

3.2.2. Nitrogen

Nitrogen appears in several chemical species or compartments:

- $\text{NO}_3^-$.
- In living and dead phytoplankton.
- As Kjeldahl nitrogen loadings.

In the model we had to separate organic N from loadings and organic N from phytoplankton, because they react differently. They are measured together as Kjeldahl N, which is defined as the difference between total nitrogen and nitrate plus NH$_3$. NH$_3$ is not separated from nitrate in the model because the NH$_3$ concentration is low (0.3 in winter and 0.1 in summer) compared to nitrate to which it is
converted. Thus we make no significant error if we immediately add this fraction to nitrate, unless there are important fluxes from NH\textsubscript{3} which would disturb the nitrogen mass balance.

One of the possible nitrogen fluxes is evaporation of NH\textsubscript{3}, but determining its rate requires knowledge of the saturation concentration in water, which of course depends on the concentration in the air. But we could not find accurate data for the NH\textsubscript{3} evaporation rate, probably because it is highly site specific. Therefore we assumed a saturation concentration of 0.0. Since NH\textsubscript{3} is in equilibrium with NH\textsubscript{4}\textsuperscript{+}, which will not evaporate, the pH-dependent ratio of NH\textsubscript{3} to NH\textsubscript{4}\textsuperscript{+} also determines the flux rate:

<table>
<thead>
<tr>
<th>pH</th>
<th>NH\textsubscript{4}\textsuperscript{+}/NH\textsubscript{3}</th>
</tr>
</thead>
<tbody>
<tr>
<td>7</td>
<td>182</td>
</tr>
<tr>
<td>8</td>
<td>18.2</td>
</tr>
<tr>
<td>9</td>
<td>1.82</td>
</tr>
</tbody>
</table>

Weiler [3.21] has experimentally established an equation for the transfer rate of NH\textsubscript{3} to the air:

\[
\text{Flux} = \text{DELT}\times k\times xmf(\text{NH}_3) \quad (3.8)
\]

where:

\[
k = 0.0242 + 0.0079\times\text{wind} \quad (3.9)
\]

A mean wind velocity of 5 m/s would result in a transport coefficient of 6 cm/day, which is the same value used for the denitrification reaction. Hence in case there is a low NH\textsubscript{3} concentration, we could overestimate the loss of nitrogen, because our denitrification flux is too high. However, this error is completely irrelevant compared to the uncertainties in the denitrification rate constants.

In the nutrient model we must calibrate three nitrogen fluxes: (1) denitrification of nitrate, (2) nitrification of Kjeldahl N loadings, and (3) nitrogen from the bottom.

3.2.2.1. Denitrification. Denitrification is possible only at a redox potential below 300 mV, hence in anaerobic systems. It has been suggested that these low redox potentials might exist locally in the bulk water in suspended flocs. If there is still a reasonable amount of oxygen in the bulk water, we can compute, however, that anaerobic conditions in the flocs can occur only if they have a
diameter of several millimeters. Otherwise the oxygen gradient and the resulting flux of oxygen would have to be so high that the mineralization rates would become unrealistic.

If denitrification reactions are to continue, therefore, nitrate must be transported to the anaerobic sediment. If the rate of the reaction exceeds the transport rate, the latter is limiting to the overall denitrification rate and a gradient of nitrate will exist in the denitrification zone. If, however, the reaction rate itself is the slower of the two steps, more nitrate will be transported to the denitrification zone than actually used and it will accumulate.

Experimental results in Ref. 3.22 suggest that the measured disappearance of nitrate still depends on the concentration at levels of 25 ppm. Because in Dutch lakes observed nitrate concentrations are always below 10 ppm, we have assumed transport of nitrate to be the rate-limiting step. Thus the denitrification rate will be proportional to the nitrate concentration in the bulk water, with the constant of proportionality to be found by calibration.

3.2.2.2. Nitrogen from Kjeldahl Loadings (BODN). Two factors contribute to Kjeldahl nitrogen in waters: (1) Kjeldahl N from phytoplankton developed in the bulk water, and (2) Kjeldahl N from external loadings. The decay constants for the nitrification of the latter were estimated from literature data.

With a rather high residence time of about 150 days, about 80 to 90 percent of these loadings are nitrified to nitrate at a concentration in the inlet water of about 1.5 to 2 ppm N, except in winter. Thus these loadings have the same effect as nitrate loadings. Moreover, since nitrate is subject to denitrification and Kjeldahl N is not, loadings of Kjeldahl N rather than nitrate provide a high available amount of nitrogen for phytoplankton.

3.2.2.3. Nitrogen from the Bottom. It may be concluded from Table 3.1 that significant bottom fluxes of nitrogen are in the form of NH₃. Since NH₃ is not included in the bulk water system, we have used a nitrate flux to the bulk water instead. We may conclude from the figures that this flux can have a substantial contribution to the amount of nitrate in the bulk water when the nitrate concentration is low. It may even become higher than the loss of nitrate due to denitrification. It would be rather difficult to change the magnitude of the denitrification and NH₃ flux and still obtain essentially the same fit for the nitrate concentration.

3.2.3. Phosphorus

Because the main component of national policy is to decrease the available amount of phosphorus, predicting fluxes of this nutrient is extremely important. Unfortunately, it is also very difficult. Besides uptake and release of phosphorus by phytoplankton, the following processes are of importance:
-32-

- Phosphorus adsorption.
- Fluxes of adsorbed, inorganic phosphorus to the bottom (sedimentation).
- Fluxes of phosphorus from the bottom, both in dissolved and suspended form (see Sec. 3.1).

The $\text{PO}_4^{3-}$ ion can be adsorbed to the surface of solid materials, mainly clay, and to oxides and hydroxides of aluminum and iron. It also has been shown that $\text{PO}_4^{3-}$ can be adsorbed to freshly precipitated CaCO$_3$. The amount which is adsorbed depends on pH and usually decreases as pH rises:

$$\text{adsorbens-P} + \text{OH}^- \rightarrow \text{adsorbens-OH} + \text{P} \quad (3.10)$$

Adsorption to CaCO$_3$ is an exception, however, because more of it is formed when the pH rises.

Adsorption reactions per se do not enhance fluxes, if the adsorbing material remains in suspension. Sedimentation is necessary to remove phosphorus from the bulk water system. The inlet water of the system to which the model is calibrated (Ring 2 of Grote Rug) is dosed with approximately 4 mg Al/l. In this particular example, adsorption to Al(OH)$_3$ will be much larger than all other adsorption processes.

Jacobsen [3.23] shows that adsorption of phosphorus to suspended material is a rapid process. Preliminary results from experiments currently performed by DHL confirm Jacobsen's conclusion but also show that desorption is a very slow process. No desorption was detectable two days after phosphorus-bearing suspended material from the Rijn had been brought into contact with phosphorus-poor water. This could imply that resuspension will only contribute marginally to the phosphorus flux into the overlying water.

3.2.4. Silicon

From a chemical point of view modeling silicon is hardly interesting. Dissolved silicon concentrations in the water column are only determined by: (1) uptake and release by phytoplankton, (2) inflow and outflow, and (3) release from the bottom. Unlike nitrogen, silicon does not affect other chemical variables.
3.3. PHYTOPLANKTON

CHARON deals with phytoplankton in the same way as the oxygen model OXYMOD does. We added interactions only for the various nutrients and carbon. Coefficient values are derived from the stoichiometric coefficients and the carbon to chlorophyll ratios of BLOOM II. Constants for processes such as mineralization are identical to the ones used in OXYMOD.

REFERENCES


Chapter 4

FORMULATION OF THE NUTRIENT MODEL

CHARON consists of two modules: an equilibrium module and a slow reaction module. Rapid processes with time constants less than one day are described in the equilibrium module. Slow processes with time constants on the order of days or weeks are described in the slow reaction module. The distinction between these two subsystems allows us to select an optimum solution for both of them.

4.1. THE EQUILIBRIUM MODULE

The equilibrium module is based on Rand’s chemical equilibrium model CHEMQ, which is described in Refs. 4.1, 4.2, and 4.3. The original program was modified in several ways to make it more suitable for modeling natural water chemistry. All of these changes concern the dependence of the Gibbs free energy parameters on temperature and ionic strength.

4.1.1. Equilibrium Chemical Systems

In this section we shall recall some of the basic principles of the equilibrium model. As the original Rand literature is mainly written for mathematicians, we shall limit the present discussion to chemical aspects.

The model is based on two central concepts: mass balance equations and minimizing Gibbs free energy. The mass balances are a set of linear equations, which state that each chemical element in the system (called a component) must be conserved.

Define the following symbols:

\[ x_j = \text{the number of moles of species } j, \]
\[ a_{i,j} = \text{the stoichiometric coefficient of species } j \text{ for component } i, \]
\[ b_i = \text{the number of moles of component } i. \]

We can now write:

\[ \sum_{j} x_j a_{i,j} = b_i \quad \text{(for every } i) \quad (4.1) \]
These equations are called the mass balance equations. Alone they are insufficient to specify the equilibrium composition of a chemical system, since the number of species in the system is usually higher than the number of mass balance equations. To specify the equilibrium composition, we introduce the Gibbs free energy function. Define:

\[ c_j = \text{the Gibbs free energy parameter of species } j, \]
\[ xmf_j = \text{the mole fraction of species } j, \]
\[ F = \text{the Gibbs free energy of the system.} \]

Then the Gibbs free energy function can be written as:

\[
F = \text{SUM}(x_j \ast (c_j + \log(xmf_j))) \tag{4.2}
\]

To be at equilibrium, the Gibbs free energy of a chemical system should be minimal.

An example will help to clarify matters. In pure water three species may be distinguished, namely \( \text{H}^+ \), \( \text{OH}^- \), and \( \text{H}_2\text{O} \), called "hydrogen ion," "hydroxyl ion," and "water," respectively. The three species can react by:

\[
\text{H}^+ + \text{OH}^- \rightarrow \text{H}_2\text{O} \tag{4.3}
\]

That is, one hydrogen ion and one hydroxyl ion can combine to form one molecule of water. The reverse reaction is also possible, whereby a molecule of water splits into its constituent hydrogen and hydroxyl ions.

We can specify two mass balance equations for this system which require that the amounts of \( \text{H}^+ \) and \( \text{OH}^- \) be conserved (these are our components). That is, every individual \( \text{H}^+ \) ion originally in the system must remain there, either in the form of the species \( \text{H}^+ \) or joined with an ion of \( \text{OH}^- \) in a molecule of water. There is a similar conservation requirement for \( \text{OH}^- \). The mass balance equations can thus be written as:
\[ \text{H}^+ + \text{H}_2\text{O} = \text{(amount of H}^+ \text{ component)} \]
\[ \text{OH}^- + \text{H}_2\text{O} = \text{(amount of OH}^- \text{ component)} \]

In this example we have two components, \( \text{H}^+ \) and \( \text{OH}^- \), and three species, \( \text{H}^+ \), \( \text{OH}^- \), and \( \text{H}_2\text{O} \). Each stoichiometric coefficient is 1.0 or 0.0. As with most chemical systems, the mass balance equations of our example do not completely determine the equilibrium composition. Given any composition (i.e., a specification of the amounts of all species) that satisfies both mass balance equations, if any \( \text{H}_2\text{O} \) is present, then some can dissociate into its constituent \( \text{H}^+ \) and \( \text{OH}^- \) ions; and if both \( \text{H}^+ \) and \( \text{OH}^- \) are present, equal amounts of each can combine to form \( \text{H}_2\text{O} \). These reactions proceed very rapidly in either direction--within less than a second--but our interest lies not in these rapid reactions but in the final state of the system, when it has achieved equilibrium.

This equilibrium can be computed with the laws of chemical thermodynamics, which state that the equilibrium composition is determined by two processes:

- Minimizing energy (enthalpy). This is the amount of energy that can be measured when a reaction proceeds as release or uptake of heat (symbol \( H \)).
- Maximizing entropy (disorder: symbol \( S \)).

Usually both processes work in opposite directions; hence a system with a maximum of disorder cannot have a minimum enthalpy. Both processes are related in a new state variable called Gibbs free energy (symbol \( F \)):

\[ F = H - TS \quad (4.4) \]

where \( T \) is the absolute temperature in degrees Kelvin.

As a result of both processes, the Gibbs free energy is minimized. The equilibrium state of a chemical system is characterized by the lowest possible Gibbs free energy. The Gibbs free energy of an entire system is the sum of the Gibbs free energies of all species in the system.
In our example of a drop of pure water, there are three chemical species, $H^+$, $OH^-$, and $H_2O$, which can react according to Eq. (4.3). Thermodynamic theory states that:

- The maximum order and minimum energy are achieved when only $H_2O$ is present, since this reaction is exothermic (it releases energy).
- The minimum order and maximum energy are achieved when only $H^+$ and $OH^-$ are present, which is the state with the greatest number of molecules, hence disorder.
- The equilibrium situation is between these extreme conditions.
- The contribution to the Gibbs free energy of each species is given by:

$$F_j = x_j^*c_j + RT^*\log(xmf_j)$$

$$= x_j^*c_j + x_j^*RT^*\log(xmf_j) \quad (4.5)$$

where $F_j$ = Contribution to Gibbs free energy of species $j$,
$x_j$ = Number of molecules of species $j$,
$c_j$ = Relative Gibbs free energy of species $j$,
$R$ = Gas constant,
$T$ = Absolute temperature,
$xmf_j$ = Mole fraction of species $j$ (the number of moles of a species divided by the total number of moles in the phase. Mole fraction is a dimensionless concentration unit).

Because $c_j$ is multiplied by $x_j$, any species with a low $c_j$ value (a large negative number), of which a considerable amount is present, will strongly reduce $F_j$. If, however, $x_j$ is very small, $xmf_j$ will be small, too: $\log(xmf_j)$ must be a large negative number and contribute strongly to a low value of $F_j$.

In the example of $H_2O$ with $H^+$ and $OH^-$, $c(H_2O)$ is low relative to $c(H^+)$ and $c(OH^-)$. Hence the minimum is obtained at a relatively large amount of $x(H_2O)$ and low amounts of $x(H^+)$ and $x(OH^-)$. The first term in Eq. (4.5), $x_j^*c_j$, is the contribution of the enthalpy;
the second, \( RT x_j^* \log(xmf_j) \), the contribution of the entropy. In the formulation of the model, Eq. (4.5) is divided by \( RT \), hence:

\[
\frac{F_j}{RT} = \frac{x_j^* c_j}{RT} + x_j^* \log(xmf_j)
\]  

(4.6)

In the remainder of this text we shall simply write:

\[
F_j = x_j^* (c_j + \log(xmf_j))
\]  

(4.7)

where both \( F_j \) and \( c_j \) are the original Gibbs free energy parameters divided by \( RT \). The values for all thermodynamic parameters have been obtained from Refs. 4.6 and 4.7.

4.1.2. Nonideal Chemical Equilibria

Technically, the dependence of the Gibbs free energy on the log of the mole fraction shown in Eq. (4.6) is incorrect, as \( F \) is related to the activity instead of the mole fraction. In so-called ideal chemical systems, the activity equals the mole fraction, but most systems are at least slightly nonideal. For nonideal systems, a more complex expression is needed for Gibbs free energy. We shall not discuss the precise nature of activity, which is excellently discussed in several textbooks, e.g., Ref. 4.4, but will confine ourselves to the solution of this problem. The activity is related to the mole fraction as:

\[
\text{Activity} = \text{Activity coefficient} \times \text{Mole fraction.}
\]  

(4.8)

Define the following symbols:

- \( g_j \) = Activity coefficient for species \( j \),
- \( A, B \) = Temperature dependent parameters,
- \( Z_j \) = Charge of species \( j \) squared,
- \( I \) = Ionic strength,
- \( b \) = A constant,
- \( a_j \) = Diameter of the charged species.
For natural waters, the activity coefficient can be calculated with the theoretical equation of Debye-Hückel for charged species:

\[
\log g_j = \frac{-A^*Z_j^*\text{SQRT}(I)}{1 + B^*a_j^*\text{SQRT}(I)} + b^*I \tag{4.9}
\]

I is calculated from:

\[
I = 0.5 \times \text{SUM}(x_j \times Z_j) \tag{4.10}
\]

For uncharged, dissolved species, the equation of Helgeson [4.5] is used:

\[
\log(g_j) = C^*I \tag{4.11}
\]

where C is a constant. For the log of the activity of species j, we can now write:

\[
\log(\text{activity}_j) = \log(g_j^*xmf_j) = \log(g_j) + \log(xmf_j) \tag{4.12}
\]

or using Eq. (4.7):

\[
F_j = x_j \ast (c2_j + \log(xmf_j)) \tag{4.13}
\]

where:

\[
c2_j = c_j + \log(g_j) \tag{4.14}
\]

The reader should recognize the illusory character of Eq. (4.14). It appears that we have been able to represent the nonideal aspects of the system merely by changing the relative Gibbs free energies, c_j, of the various species. Each c_j has simply been augmented by the logarithm of the activity coefficient. What must be recognized is that the activity coefficient is not constant, but depends on the
composition of the chemical system. Thus, we calculate the activity coefficients corresponding to our best guess at the equilibrium composition, adjust the $c_j$'s according to Eq. (4.14), and then find the composition that minimizes the modified ideal form of the Gibbs function subject to the mass balance constraints. But if the composition so calculated differs from the composition for which the activity coefficients were calculated—e.g., if the ionic strength $I$ has changed—then the $c_j$'s must be recalculated and the newly modified Gibbs function reminimized. Fortunately, this process converges after only a few iterations.

4.1.3. Temperature

Gibbs free energies are usually given at the so-called standard state (25 deg C, 1 atmosphere, zero ionic strength). Hence other values should be used for different temperatures. According to thermodynamic theories we may write:

$$\frac{d(F_j/T)}{dT} = -\frac{H_j}{T^2} \quad (4.15)$$

where $F_j = RT \ast c_j(T)$. Integrating, we find:

$$\frac{F_j(T)}{T} - \frac{F_j(T_0)}{T_0} = H_j \ast (\frac{1}{T} - \frac{1}{T_0}) \quad (4.16)$$

where $T_0 = 298.15$ deg K (25 deg C). Hence:

$$c_j(T) = c_j(T_0) + H_j \ast (T_0 - T)/(RT_0) \quad (4.17)$$

If the enthalpy of a species $H_j$ is known, we can calculate the value of $c_j$ at any temperature. In the previous derivation we have assumed $H_j$ to be independent of $T$. This is not completely true, but deviations are small over the temperature range considered.

In the applications considered here, the temperature of the chemical system is always known. Before calculating its equilibrium state,
Eq. (4.17) is used to calculate $c_j$'s for each species from their values in the standard state. Unlike the correction for nonideality, the temperature correction is independent of composition, and hence can be applied once, in advance of the calculation of the equilibrium. It is not necessary to iterate in order to converge upon the proper temperature correction.

4.2. THE SLOW REACTION MODULE

In a natural water system, many reactions occur which are not in equilibrium. Moreover, the thermodynamic laws should be applied only to closed systems, but water in a natural water system is (slowly) replaced and transport phenomena are important. Slow reactions concern us for two reasons:

- They can change the amounts of components at equilibrium; e.g., decay of BOD adds CO$_2$ to and subtracts O$_2$ from the equilibrium system.
- Several of the chemical species of importance to eutrophication are involved in slow reactions.

Therefore the equilibrium module was extended with a slow reaction module consisting of:

- A set of ordinary first order differential equations describing the rates of change of slow reactants with respect to time.
- A procedure to calculate the exchange of components with the equilibrium module to which it is coupled.

The value of one of the equilibrium variables appears on the right-hand side of several of the differential equations in the slow reaction module. Hence the solution of the differential equations depends on the equilibrium system, which in its turn depends on the solution of the differential equations.

4.2.1. Formulation of Differential Equations

In Eq. (4.18) below, we show the formulation for the differential equations of slow reactants. Although not all slow reactants are formulated in exactly the same way, the essential elements are always similar.

Define the following symbols:
SR\(_i\) = The number of moles of slow reactant \(i\),

\(k_i\) = A rate constant, usually a function of temperature, wind, or both,

\(AV_i\) = Either the concentration of species \(i\) as mole fraction, or the amount in moles, as calculated by the equilibrium module.

Then we can write a differential equation:

\[
\frac{d(SR_i)}{dt} = k_i \cdot AV_i(t) \quad (4.18)
\]

\(AV_i(t)\) is a function of time because, through the operation of the equilibrium module, it is a function of the components of the system, and the components change as the slow reactants change.

For all slow reactants, the differential equations of (4.18) are solved by the following method:

Discretization:

\[
\text{DELT}SR_i = \text{DELT}t \cdot k_i \cdot AV_i(t) \quad (4.19)
\]

Approximation for \(AV_i(t)\):

\[
AV_i(t) = 0.5(\text{AV}_i(0) + \text{AV}_i(\text{DELT}t)) \quad (4.20)
\]

Approximation for \(AV_i(\text{DELT}t)\):

\[
\text{AV}_i(\text{DELT}t) = \text{AV}_i(0) + (\text{dAV}_i/dt) \cdot \text{DELT}t \quad (4.21)
\]

From Eqs. (4.20) and (4.21):

\[
\text{AV}_i(t) = \text{AV}_i(0) + 0.5 \cdot (\text{dAV}_i/dt) \cdot \text{DELT}t \quad (4.22)
\]
To calculate \( \frac{dAV_i}{dt} \) we recall that \( AV_i \) depends on the components of the equilibrium module, and through them on the slow reactants. Thus:

\[
\frac{dAV_i}{dt} = \sum_j \frac{dAV_i}{dSR_j} \cdot \frac{dSR_j}{dt}
\]  

(4.23)

From Eqs. (4.22) and (4.23):

\[
AV_i(t) = AV_i(0) + 0.5 \cdot \left( \sum_j \frac{dSR_j}{dt} \right) \cdot \text{DELTt}
\]

For which we write:

\[
AV_i(t) = AV_i(0) + 0.5 \cdot \left( \sum_j \frac{dSR_j}{dt} \right) \cdot \text{DELTt}
\]

(4.24)

Substituting Eq. (4.24) into (4.19) gives:

\[
\frac{dAV_i}{dSR_j} \cdot \text{DELTt} = \left( k_i^{-1} \cdot (AV_i(0) + 0.5 \cdot \sum_j \frac{dSR_j}{dt}) \right) \cdot \text{DELTSR}_j
\]

(4.25)

The derivatives \( \frac{dAV_i}{dSR_j} \) can be calculated using the so-called R matrix of the equilibrium module[4.1,4.2]. Let \( xbar \) be the total number of moles in the water phase. Then take the summation-over-\( l \) components of slow reactant \( j \) and the summation-over-\( k \) components which combine to form species \( i \). Then in the case that \( AV \) is a mole fraction (symbol \( xmf \)) we can use the following equation to calculate the derivative:

\[
\frac{dxmf_i}{dSR_j} = \sum_{k,l} \left( a_{k,i}^{-1} \cdot a_{l,k} \cdot xbar \right) \cdot \frac{x_i}{l \cdot k \cdot xbar}
\]

(4.26a)
The equation is similar if $AV$ is the number of moles of a species (symbol $x$):

$$
\begin{align*}
\frac{dx_i}{dSR_j} = & \sum_{k} \left( \sum_{1,k} a_{k,1} R^{-1}_{1,k} a_{1,j} x_i \right) \\
\end{align*}
$$

(4.26b)

We can solve the system of linear equations in (4.25), which gives the change in the number of moles of all slow reactants. A slow reactant is changed when the components of which it is composed are released or taken up, e.g., a change in the $b$ vector. The derivatives are calculated from equations such as Eq. (4.26).

From the new $b$ vector we can compute a new equilibrium solution and also a value for $AV_i(DELT_t)$. This value can be compared to the prediction of $AV_i(DELT_t)$:

$$
\frac{dAV_i}{dSR_j} = AV_i(0) + \sum_j \frac{dx_i}{dSR_j} \ \ \ \ (4.27)
$$

A correction is necessary if the difference between the predicted and calculated value of $AV_i(DELT_t)$ exceeds a certain limit. Defining $AV_i(DELT_t)$ as the prediction for $AV_i$ made in the first approximation, we can rewrite Eq. (4.25) to:

$$
DELT_t x_{k_i}^{*} = \frac{AV_i(0) + AV_i(DELT_t)}{2.0}
$$

If this prediction is incorrect, or if the calculated value for $AV_i$ is significantly different from the predicted value, then the calculated value of $DELTSR_i$ is also incorrect. Denoting $DELTSR_i^*$ and $AV_i^*$ as the actual values, we may write:

$$
DELT_t x_{k_i}^{*}
$$
\[ \text{DELT}_i^* = \frac{\text{AV}_i(0) + \text{AV}_i^*(\text{DELTt})}{2.0} \]  

(4.28)

and subtracting Eq. (4.28) from Eq. (4.27), we obtain:

\[ \text{DELT}_i^* \text{k}_i \]

\[ \text{DELT}_i^* - \text{DELT}_i = \frac{\text{AV}_i^*(\text{DELTt}) - \text{AV}_i(\text{DELTt})}{2.0} \]  

(4.29)

\[ \text{DELT}_i^* - \text{DELT}_i \] is the necessary correction on the change in slow reactant. This correction leads to a new change in slow reactant, hence:

\[ \frac{d(\text{AV}_i)}{d(\text{SR}_k)} \]

\[ \text{AV}_i^*(\text{DELTt}) = \text{AV}_i(\text{DELTt}) + \text{SUM} \frac{\text{DELT}_k^* - \text{DELT}_k}{\text{SR}_k} \]  

(4.30)

where \( \text{AV}_i^* \) is the value from the equilibrium calculation of the first approximation. Eqs. (4.29) and (4.30) can be combined into a new set of equations:

\[ \text{DELT}_i^* \text{k}_i \]

\[ \text{DELT}_i^* - \text{DELT}_i = \frac{\text{AV}_i(\text{DELTt})}{2.0} \]

\[ \frac{d(\text{AV}_i)}{d(\text{SR}_k)} \]

\[ + \text{SUM} \frac{\text{DELT}_k^* - \text{DELT}_k - \text{AV}_i(\text{DELTt})}{\text{SR}_k} \]  

(4.31)

These can be solved again, giving a new approximation for the value of \( \text{DELT}_i^* \). The corrector calculation can be repeated as often as necessary. Actually the model does not compare all calculated
mole fractions with the predictions to decide whether a new corrector step is necessary. It compares the differences between quantities called Lagrange multipliers or partial Gibbs free energies of components before and after the equilibrium calculation, and stops iterating when these differences have become smaller than a certain limit.

4.2.2. Types of Slow Reactions

Presently there are five types of slow reactions in the model. Defining $AV_i$(sat) as the value of $AV$ in a system where the slow reactants are in equilibrium, we can write the next general equation for the first three types of slow reactions:

$$\frac{d(SR_i)}{dt} = k_i * (AV_i(t) - AV_i(sat))$$

(4.32)

The formulations for the fourth and fifth type differ to some extent:

$$\frac{d(AV_i)}{dt} = k_i * AV_i(t) + \text{constant}_i$$

(4.33)

4.2.2.1. Type One Slow Reactions. In the type one reaction, $AV$ is the ion activity product of a mineral, and $AV_i$(sat) is its solubility product. Since this type is unimportant for surface water calculations with the current version of the model, it will not be discussed.

4.2.2.2. Type Two Slow Reactions. In the type two reaction $AV$ is the dissolved species concentration with which the slow reactant reacts; e.g., the $O_2$ concentration in the water of slow reactant $O_{2,\text{air}}$ and $AV_i$(sat) is the saturation concentration or the concentration which would exist at equilibrium.

4.2.2.3. Type Three Slow Reactions. In the type three reaction, $AV$ is the concentration of some species, and $AV_i$(sat) is zero; e.g., for
denitrification AV is the concentration of $\text{NO}_3^-$, and the slow reactant is a species consisting of $\text{HNO}_3$. For sedimentation of calcite, AV is the concentration of $\text{CaCO}_3$.

4.2.2.4. Type Four Slow Reactions. In the type four reaction, the slow reactant is not an actual species but a component, and $AV_i$ is the amount of component in the system. This type is applied to describe input and output of components to and from the equilibrium system.

4.2.2.5. Type Five Slow Reactions. In the type five reaction, AV is the amount of the slow reactant itself (first order) or is 1 (zeroth order), and the constant can have any value.

4.3. COMPONENTS SELECTED

In the model we have included each component that is important to describe the behavior of nutrients in natural lakes. Hence we have included the major constituents of algae: phosphorus (P), nitrogen (N), silicon (Si), and carbon (C). Each of these elements appears in several chemical species, some of which contain other elements as well. Therefore we have also included some other elements, which are not directly important to algae, but influence the behavior of nutrients in the water. These are oxygen (O), hydrogen (H), calcium (Ca), and every element related to suspended adsorbing material. Other relatively abundant elements such as chloride (Cl) and sodium (Na) are not included since they do not influence the processes we are investigating. These elements do affect the ionic strength, but this effect was computed for "mean" Rijn water conditions and permanently included as an input to the model. Although the ionic strength is not constant in Dutch lakes, the variations in activity coefficients caused by it are small in comparison with the accuracy of data. Hence we concluded that we could use a mean activity coefficient in all calculations.

4.4. CHEMICAL SPECIES SELECTED

Of all the chemical species that may be composed of the components in the model, we have selected those which appear in relatively large quantities under natural conditions. For example, we have not included $\text{H}_3\text{PO}_4$, although it is a species of P, because it is only important at much lower values of the pH than are ever observed in Dutch eutrophic lakes.
4.5. PROCESSES INCLUDED IN THE SLOW REACTION MODULE

We must include all slow processes in the model which are directly or indirectly important to the variables we are interested in, such as nutrients for phytoplankton. Thus sedimentation of inorganic adsorbed P is clearly important because it alters the amount of P available to phytoplankton. As another example, the exchange of CO₂ between air and water has to be included because the amount of CO₂ has a strong influence on the pH and hence on the adsorption of P.

We have also included several other processes which may not be directly important, but are affected by some of the processes included in the model. For example, it is not necessary to model oxygen, but it is simple and provides additional information which may be used in a calibration.

We have included the following slow processes in the model:

• Exchange of CO₂ and O₂ between air and water. Exchanges of N₂ or NH₃ are not included for reasons explained in Chap. 3.
• Exchanges between bottom and water:
  - Of all selected components.
  - Sedimentation of suspended materials.
  - Denitrification.
• Decay of unstable species:
  - Decay of BODC (carbon).
  - Decay of BODN (nitrogen).
• Loadings to the system and outflow of all selected components.
• Uptake and release of components by biological processes (phytoplankton blooms).

4.6. INPUTS TO THE MODEL

CHARON requires:

• Formulation of the equilibrium system in terms of stoichiometric coefficients, Gibbs free energy parameters, enthalpy parameters, ion sizes, etc.
• Formulation of the slow reaction system in terms of type of
reactants, rate constants, etc.

- Specification of the initial conditions.
- Time series giving weekly values for:
  - Temperature.
  - Wind velocity.
  - Loadings of all selected components, usually given as measurements of concentrations.
  - Loadings of all species involved in slow reactions such as BOD.
  - Exchange of components between biological and chemical processes, in terms of measured chlorophyll or computed carbon concentrations.

NOTES

1. The Lagrange multipliers are the results of an intermediate calculation in the equilibrium module. They have an intimate mathematical relation with the mole fractions of the species. For more information, see Ref. 4.1, where they are called PIE's.

REFERENCES

Chapter 5
CALIBRATION AND VALIDATION OF THE NUTRIENT MODEL

5.1. THE CALIBRATION PROCESS

As mentioned earlier, the values for most of the rate constants were taken from previous studies. The only rate constants determined by calibration are: (1) transport rates for CO$_2$ and O$_2$ from the atmosphere, (2) sedimentation rates for suspended material, and (3) the dispersion coefficient of the components from the bottom fluxes.

5.1.1. CO$_2$ and O$_2$ Transport Coefficients

We have adjusted the CO$_2$ transport rates of the model until the computed values of the pH matched the measurements sufficiently well. By this calibration procedure we obtained a value of 0.07 for CO$_2$ going into the water and a value of 0.04 for CO$_2$ transport toward the air. A value of 0.045 is used for transport of oxygen in both directions.

5.1.2. Sedimentation of Suspended Material

For the adsorbed phosphate species, we have selected values for the free energy parameters and stoichiometric coefficients such that pH behaves similarly, as reported by Van der Meulen [5.1]. Sedimentation rate constants for all suspended materials were selected and varied until a good fit was obtained. The sedimentation constant was taken as 60 cm/day.

5.1.3. Dispersion Coefficient

A good fit to the data could be obtained only by calibration of the dispersion rates from the bottom. From the concentration gradients shown in Table 3.1, we could compute the rates of the fluxes, assuming a dispersion coefficient equal to molecular diffusion coefficients. It appeared we had to use much higher dispersion coefficients, as shown in Table 5.1.

The numbers in Table 5.1 are computed with the assumption that the concentrations have been measured at a depth of 1 cm in the oxygen layer; the total depth of this layer is about 2 cm in this case. Thus the gradient is the concentration difference between the oxygen layer
Table 5.1

REQUIRED MULTIPLICATION FACTOR OF MOLECULAR DIFFUSION RATES TO CALIBRATE THE NUTRIENT MODEL

<table>
<thead>
<tr>
<th>Species</th>
<th>Multiplication Factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>PO$_4^{3-}$</td>
<td>12.5</td>
</tr>
<tr>
<td>NH$_3$</td>
<td>5.0</td>
</tr>
<tr>
<td>H$_4$SiO$_4$</td>
<td>4.0</td>
</tr>
<tr>
<td>NO$_3^-$</td>
<td>6.2</td>
</tr>
<tr>
<td>CO$_2$</td>
<td>8.0</td>
</tr>
</tbody>
</table>

and overlying water divided by 1 cm. Obviously this approximation is rather rough, and it is quite possible that higher or lower gradients exist for some species at the exchange interface. If indeed mixing and bioturbation prevail under current conditions, we should find the same dispersion coefficients for all species. But we cannot calculate the gradient accurately because, as we have mentioned earlier, the actual depth of the measurements is unknown. Moreover, reactions within the oxidized layer can entirely change the picture.

Fluxes from the interstitial water to the bulk water can also be caused by resuspension, which is another possible source of errors in the calculated flux rates. On the average the sedimentation flux will be equal to the resuspension flux. But, for instance, adsorbed PO$_4^{3-}$ can desorb after resuspension, because the pH in the bulk water rises. This resuspension flux might make a substantial contribution to the total flux. The importance of resuspension and following desorption may, however, be seriously doubted, as recent investigations for WABASIM have shown (see also Sec. 3.2.3).

5.1.4. Comments on Calibration

We have selected the smallest possible value for the flux rate of phosphorus which enables the model to fit observations. The model can also be calibrated with smaller phosphorus and sedimentation flux rates for a considerable part of the year, but not for the period around week 24 in the calibration data, when a large phytoplankton bloom occurred. During this bloom sedimentation became effectively zero, due to a high pH and low amount of total available phosphorus. Because an increase in total phosphorus was observed, it must have been released by the bottom. We had to assume a dispersion coefficient for orthophosphate in the interstitial water two times higher than for other bottom fluxes, which is not unrealistic. As already pointed out, the gradients cannot be determined exactly.
5.2. CALIBRATION RESULTS

With the above as the calibrated parameter values, we found (as shown in Table 5.2) that almost one-fourth of the phosphorus available to phytoplankton in Grote Rug, Ring 2, 1977 (our calibration case), came from normal fluxes from the bottom.

Table 5.2

<table>
<thead>
<tr>
<th></th>
<th>Nitrogen Flux</th>
<th>Phosphorus Flux</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loadings</td>
<td>71.5 84%</td>
<td>6.50 77%</td>
</tr>
<tr>
<td>Flux from the bottom</td>
<td>13.1 16%</td>
<td>1.94 23%</td>
</tr>
<tr>
<td>Total input</td>
<td>84.6 100%</td>
<td>8.44 100%</td>
</tr>
<tr>
<td>Outflow</td>
<td>36.2 43%</td>
<td>0.79 9%</td>
</tr>
<tr>
<td>Flux to the bottom</td>
<td>27.6 33%</td>
<td>6.35 75%</td>
</tr>
<tr>
<td>Organic sedimentation</td>
<td>20.8 24%</td>
<td>1.30 16%</td>
</tr>
<tr>
<td>Total output</td>
<td>84.6 100%</td>
<td>8.44 100%</td>
</tr>
</tbody>
</table>

Fluxes to the bottom are denitrification for nitrogen and inorganic sedimentation for phosphorus. All fluxes except loadings have been computed by the model.

Figures 5.1 through 5.8 show annual time series of computed and measured variables. From the results in these figures we may draw the following conclusions:

The pH is clearly dependent on changes in biomass levels. When biomass increases, pH is high (week 20 to 25), but when biomass decreases, it is low. Usually pH remains low for some weeks until most of the detritus which was produced is mineralized. The same trend may be observed for oxygen.

Total phosphorus follows more or less the same trend as chlorophyll. Only when phytoplankton biomass is extremely low does the phosphorus level remain relatively high. A possible explanation could be that low biomass is accompanied by low values of the pH, which favors adsorption of phosphorus to suspended material. This means that a relatively high fraction of total P is inorganic.

Increased adsorption at low values of the pH can also be observed in Fig. 5.7, which shows the phosphorus load to the system and its sedimentation. Although there is considerable variation in the flux rates due to the irregular inlet regime, it is obvious that:
Fig. 5.1--Observed chlorophyll concentration in Ring 2, 1977
Fig. 5.4—Measured and computed total P in Ring 2, 1977
Fig. 5.6—Computed nitrogen fluxes in Ring 2, 1977

* = BOTTOM N
+ = DENITRIF

NITROG. FLUXES (GR/M2.WEEK) 1977 AL Dosed
Fig. 5.7--P fluxes in Ring 2, 1977, with measured loadings and computed sedimentation
They are rather well correlated, which means that most of the loading is removed to the sediment after a short period of time.

- Sedimentation of phosphorus is lower than the load between week 15 and 24, which causes an increase in the total phosphorus concentration of the water.
- Sedimentation of phosphorus is much higher than the load to the system after week 24.
- Low sedimentation rates coincide with a high pH, high sedimentation rates with a low pH.

Fig. 5.6 shows the nitrogen fluxes. It can be observed that the denitrification rate is much higher than the load from the bottom near the beginning and end of the year, when nitrate concentrations are relatively high. However, in the middle of the year the bottom load exceeds the rate of denitrification. This last result is generally true: if concentrations in the bulk water are low, fluxes from the bottom become more important.

We must make an additional remark about Fig. 5.8 for silicon. Measured data of $\text{H}_4\text{SiO}_4$ show a drop in concentration between week 30 and week 40, which is not reproduced by the model. As uptake by diatoms is the only silicon-affecting process and we have used a zero concentration for diatoms this part of the year, the model obviously could not reproduce this phenomenon. However, in a later coupled version of the models BLOOM II and CHARON, diatoms were indeed predicted and silicon decreased during this period.

The sequence of events which enables the coupled model to produce diatoms in the middle of summer is an interesting illustration of the interaction of biological and chemical processes:

- The value of the pH becomes extremely low when the bloom suddenly starts to collapse after week 24, because mineralization of detritus produces an enormous amount of $\text{CO}_2$.
- Because pH is so low, most of inorganic phosphorus is adsorbed.
- Because little phosphorus is available, total biomass is extremely low.
- Because total biomass is low, the total extinction is also low, which favors growth of diatoms.

5.3. VALIDATION

We have tried to validate the nutrient model against the data of the undosed control Ring 3 of Grote Rug for the year 1976. Figures 5.9 through 5.11 clearly demonstrate that the validation completely fails. The agreement is rather good in the first half of the year, but after week 30, computed and observed numbers do not match at all.
Fig. 5.9--Measured and computed total P in Ring 3, 1976
Fig. 5.10—Measured and computed Si in Ring 3, 1976
A careful analysis of all results leads to the hypothesis that an explosive flux occurred. There are several indications to support this hypothesis:

1. Nitrate concentrations are 0.0 in week 30.
2. Not only phosphorus, but also silicon concentrations increase substantially.
3. Measured concentrations of sulfate are lower than computed after week 30 suggesting consumption of sulfate by the reduced bottom. Remember that we had assumed sulfate to be conservative in the model.
4. There is no significant difference between computed and measured nitrate levels, perhaps because there is increased flux of NH$_3$ from the bottom followed by nitrification in the water, and an enhanced denitrification at the bottom.

REFERENCE

Chapter 6
INVESTIGATIONS FOR PAWN LAKES

In June 1979 insufficient data were available to apply the nutrient model CHARON to each of the PAWN lakes. Therefore we decided only to investigate the possibilities of explosive fluxes. For that purpose the extremely warm year 1976 was excellent: mineralization rates were relatively high, and the bottom was much more likely to become reduced than in an average year.

As pointed out earlier, for most species it is difficult to determine various flux rates, because these species can participate in several chemical reactions. Silicon, however, is an exception and may therefore be used as a tracer. As stated in Sec. 3.2.4, it is affected only by loadings, sedimentation of organic Si, uptake by phytoplankton, and bottom fluxes. Any significant increase in the silicon concentration in summer must be due to loading from the bottom because external loadings are known and are always very low in summer. Moreover, silicon behavior is influenced very little by uptake and possible sedimentation of diatoms, because these are hardly present at the high extinction levels which are typical for the summer period in many eutrophic Dutch lakes.

Table 6.1
MONTHLY AVERAGE VALUES OF SILICON IN RIJN AND IJSSELMEER WATER FOR THE PERIOD 1976-1978

<table>
<thead>
<tr>
<th>Month</th>
<th>Silicon Concentration (mg/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Rijn</td>
</tr>
<tr>
<td>January</td>
<td>2.47</td>
</tr>
<tr>
<td>February</td>
<td>2.34</td>
</tr>
<tr>
<td>March</td>
<td>2.14</td>
</tr>
<tr>
<td>April</td>
<td>1.45</td>
</tr>
<tr>
<td>May</td>
<td>1.03</td>
</tr>
<tr>
<td>June</td>
<td>0.96</td>
</tr>
<tr>
<td>July</td>
<td>0.84</td>
</tr>
<tr>
<td>August</td>
<td>0.79</td>
</tr>
<tr>
<td>September</td>
<td>1.09</td>
</tr>
<tr>
<td>October</td>
<td>1.49</td>
</tr>
<tr>
<td>November</td>
<td>2.04</td>
</tr>
<tr>
<td>December</td>
<td>2.33</td>
</tr>
</tbody>
</table>

It may be concluded from Table 6.1 that the input of silicon to the IJsselmeer can have little effect on its concentration in summer, but there is nevertheless a considerable increase in the silica
concentration in the lake in July and August. In August, in fact, silica levels in the IJsselmeer are even higher than in the Rijn.

Figures 6.1 through 6.3 clearly show the occurrence of explosive fluxes from the bottom of the IJsselmeer in 1976. Similarly, abrupt midsummer rises in silica in the Veluwemeer, Wolderwijd, and Gooi/Eemmeer indicate that explosive nutrient fluxes occurred from their bottoms in 1976 as well. Furthermore, in the highly eutrophic Veluwemeer and Wolderwijd, explosive fluxes are indicated in mid-June of 1975, although no such fluxes occurred in the less eutrophic IJsselmeer in 1975.

We may draw the following conclusions from our studies with the PAWN nutrient model:

- It is very difficult to establish the relation between the external nutrient loadings and the nutrient concentrations.
- The bottom can be a very important source for nutrients.
- The entire chemical system of a lake has to be taken into account to describe nutrient behavior.

Many eutrophication models which have been proposed in the literature focus on phytoplankton and have no elaborate scheme to handle nutrients. Some of these models have been applied to Dutch lakes without much success.

Our studies indicate that the bottom—even without explosive fluxes—and the chemistry of the overlying water are of paramount importance in predicting nutrients and hence phytoplankton concentrations in Dutch waters. Fortunately this has been rather widely recognized in the last few years by several investigators and institutes, e.g., Hietjes [6.1.], and van Eck [6.2.].

Eutrophication control tactics which aim at a reduction of the external nutrient loads should be very carefully investigated before they are applied, as the following illustrates:

- A decrease in the external BOD loading to a natural water system reduces its organic content and oxygen consumption (both desirable effects) but meanwhile leads to a higher pH, which produces negative effects on the adsorption and sedimentation of phosphorus.
- Increased purification of waste water will also diminish the load of suspended material to the receiving water systems.
- When bottom fluxes are relatively high, they might provide enough nutrients to sustain high phytoplankton biomasses. It is as yet unknown how many years it will take before Dutch bottoms are depleted of phosphorus.
Fig. 6.1--Measured nitrate in IJsselmeer, 1976
Fig. 6.2--Measured Si in IJsselmeer, 1976
Fig. 6.3--Measured total P in IJsselmeer, 1976
REFERENCES


PART II: THE PHYTOPLANKTON MODEL (BLOOM II)
Chapter 7

STRUCTURE OF THE PHYTOPLANKTON MODEL BLOOM II

7.1. PURPOSE OF BLOOM II

Biologically speaking, phytoplankton are relatively primitive plantlike organisms. They require considerable amounts of nitrogen, phosphorus, solar energy, and sometimes silicon to become a nuisance. In theory each of these factors could become limiting, but the question is where and when. Also the physiological data indicate that species of phytoplankton differ greatly in nutrient requirements, efficiency of solar energy fixation (photosynthesis), and (potential) net growth rates.

To understand this complexity and predict the impacts of changing circumstances, a modified and extended version (BLOOM II) was developed from Rand's Algae Bloom Model, which was applied to the Oosterschelde sea estuary by Bigelow et al. [7.1]. Its purpose is to compute the maximum total biomass concentration of several phytoplankton species at equilibrium in a certain time period under a given set of environmental conditions. The maximum biomass, species composition, and limiting factors are computed by linear programming (LP) techniques. BLOOM II calculates maximum rather than actual biomasses for the following reasons:

1. It is assumed that those species which can produce most offspring under the prevailing conditions will outcompete the others.
2. The largest blooms are a manager's main interest.
3. Otherwise more physiological knowledge would be required.

To compute values for the environmental constraints, the model needs information on the concentrations of total available nutrients, temperature, the influx of solar radiation, and certain lake-specific characteristics (depth and turbidity). These conditions can all be determined directly or indirectly from measurements and are sufficient for the model's calibration and validation.

Under many conditions, phytoplankton species can achieve high net growth rates enabling them to double their biomass several times a week, sometimes even a day. Thus the model assumes steady states for both phytoplankton biomass and nutrient recycling with a nominal time-step of one week. In BLOOM II, therefore, succeeding time-steps are completely independent, although slowly changing environmental conditions tend to give the model's output a smooth appearance. Besides these nominal options, BLOOM II has additional options to:
1. Increase its nominal time-step during the entire year or some parts of it.
2. Solve the equations for nutrient recycling dynamically, maintaining the steady state for phytoplankton.
3. Include constraints for the maximum growth rates of individual species during one time-step.

In addition to the abiotic conditions, zooplankton biomasses may be provided to the model to compute losses due to grazing. For PAWN this was impossible, however, as there were no zooplankton data for any of the PAWN lakes. This is not considered to be an important problem as (1) zooplankton grazing was insignificant in most Grote Rug cases, and (2) the computed blooms for the PAWN lakes are usually dominated by species which presumably are not eaten by zooplankton, such as species of blue-green phytoplankton. Modeling zooplankton might be reconsidered if "edible" species increased significantly under new, simulated conditions.

7.2. SELECTION OF PHYTOPLANKTON SPECIES

Natural blooms often consist of assemblages of different phytoplankton species, sometimes of rather distinct groups of species such as diatoms, green, and blue-green phytoplankton. The adverse impacts of eutrophication are closely related to group and even species dominance. For example, a bloom of blue-greens is usually considered far worse than one which is dominated by diatoms or green phytoplankton. Thus it makes sense to distinguish between groups.

But particularly within the group of blue-greens still more details are required, because the impacts of a bloom of, for example, Oscillatoria agardhii are quite distinct from a bloom of Microcystis aeruginosa. If there is a bloom of the latter, the probability of a sudden collapse, hence of anaerobic conditions, seems much greater. Microcystis is also the most important blue-green species for which toxic effects have been reported by Gorham [7.2] and others. Thus it is a major advantage if a management model includes a moderately large number of species with distinct ecological characteristics. Ten different phytoplankton species, selected from various (Dutch) literature sources, are incorporated in the present version of BLOOM II (Table 7.1). Among these are the dominant species of most of the lakes considered for PAWN.
Table 7.1
NAMES OF GROUPS AND SPECIES IN BLOOM II AND ABBREVIATIONS USED IN THIS VOLUME

<table>
<thead>
<tr>
<th>Group Name</th>
<th>Species Name</th>
<th>Abbreviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diatoms</td>
<td>Asterionella</td>
<td>D1</td>
</tr>
<tr>
<td>Diatoms</td>
<td>Centric Diatoms</td>
<td>D2</td>
</tr>
<tr>
<td>Flagellates</td>
<td>Cryptomonas</td>
<td>F1</td>
</tr>
<tr>
<td>Flagellates</td>
<td>Volvox</td>
<td>F2</td>
</tr>
<tr>
<td>Greens</td>
<td>Scenedesmus</td>
<td>G1</td>
</tr>
<tr>
<td>Dinoflagellates</td>
<td>Ceratium</td>
<td>I1</td>
</tr>
<tr>
<td>Blue-greens</td>
<td>Anabaena</td>
<td>B1</td>
</tr>
<tr>
<td>Blue-greens</td>
<td>Aphanizomenon</td>
<td>B2</td>
</tr>
<tr>
<td>Blue-greens</td>
<td>Microcystis</td>
<td>B3</td>
</tr>
<tr>
<td>Blue-greens</td>
<td>Oscillatoria</td>
<td>B4</td>
</tr>
</tbody>
</table>

7.3. NUTRIENT REQUIREMENTS OF PHYTOPLANKTON SPECIES

7.3.1. Nutrient Mass Balances

To calculate how much phytoplankton biomass can be sustained, it is necessary to determine (1) the total amount of each nutrient in the water, and (2) the available and unavailable fractions as defined in Sec. 1.5.2. As indicated in Fig. 1.3, the total nutrient amounts in the water change by means of two important processes, water transport (inflow and outflow) and exchange with the bottom. Both are modeled in CHARON, but as an integrated version of this model and BLOOM II was not available for PAWN, a comparatively simple approach using measured concentrations was adopted for calibration and validation of BLOOM II.

Define the following symbols:

\[ x_j \] = the amount of phytoplankton species \( j \),
\[ a_{i,j} \] = the amount of nutrient \( i \) per unit of species \( j \):
  - the minimum stoichiometric constant,
\[ y_i \] = the amount of nutrient \( i \) which is (temporarily) unavailable, because it is incorporated in dead phytoplankton,
\[ e_i \] = the surplus of nutrient \( i \) which is directly available,
\[ b_i \] = the total readily available amount of nutrient \( i \).

For the total amount of nutrient \( i \) in the water, the following mass balance equation should hold:
\[ b_i = \sum_{j} a_{i, j} x_j + y_i + e_i \]  

(7.1)

To obtain values for the \( b_i \), we use the measured total concentrations. It is implicitly assumed that every nutrient fraction included in these measurements may become available during a time-step. Obviously the total amount in the water is an upper bound to the total available amount \( b_i \), which is in accordance with the purpose of BLOOM II.

Experience with this model and CHARON indicates that frequently a large portion of the total amount may indeed be considered available, but as will be shown later in Chap. 9, there are some important exceptions.

We assume that none of the \( b_i \) values change within a time-step: there is no net transport to or from the bottom and connecting waterways.\(^1\) This assumption had to be adopted as the model receives new total available nutrient concentrations for each time-step. Thus in the (exceptional) case when there is a large change in some or all of the concentrations, the model's biomass computations may be incorrect for that period, but in the next time-step these transports are implicitly accounted for in the new inputs of \( b_i \).

In the form of Eq. (7.1) used in Ref. 7.3, there is a term for nutrients in zooplankton. We omit this term here since no data are available on zooplankton in PAWN lakes. This is consistent with the purpose of BLOOM II: nutrients in zooplankton are included in the total measured amount, and hence the model considers them to be available, although actually they are not directly available to phytoplankton.

### 7.3.2. Nutrient Recycling

Frequently a substantial amount of nutrients is tied up in dead phytoplankton cells (detritus) and is thus temporarily unavailable to living cells. In BLOOM II, this amount is denoted by \( y_i \). Modeling detritus is difficult for two reasons: (1) there is no way to separate detritus from other fractions in a sample, and hence the computed number cannot be compared to an observation; and (2) there is considerable uncertainty about the rates at which detritus is generated and removed.

A certain amount of each species \( j \) dies during each time-step, withdrawing its nutrient content from the live phytoplankton pool. Only a fraction of this nutrient remains a part of the detritus pool, however. The rest becomes directly available to growth of new individuals, because the dead cells break apart (a process called autolysis), spilling some of their contents into the water in dissolved form. Detritus may be removed to the bottom or to the
dissolved nutrient pools at rates in proportion to its concentration. We can express this mathematically as follows. Define:

\[ M_j \]  = the mortality rate of species \( j \),
\[ q_i \]  = the fraction of dead phytoplankton cells which has to be remineralized and is not immediately released when a cell dies,
\[ u_i \]  = the remineralization rate constant of dead phytoplankton cells per day,
\[ s \]  = the sedimentation rate constant of dead phytoplankton cells per day.

Then the rate of change of the amount of nutrient \( i \) in detritus can be described by the following equation:

\[
\frac{d y_i}{dt} = \sum_j (q_i \cdot M_j \cdot q_i \cdot x_j) - u_i \cdot y_i - s \cdot y_i \quad (7.2)
\]

For each \( q_i \) we adopt a value of 0.5. Thus, 50 percent of each nutrient becomes (almost) instantaneously available when a cell dies. The nominal value for \( s \) is 0.0 since, as previously explained, BLOOM II assumes no transports to or from the water during a time-step. The remineralization rate constants \( u_i \) are computed as:

\[
u_i = 0.006 \times T \quad (P \text{ and } N) \quad (7.3a)
\]

\[
u_i = 0.025 \quad (Si) \quad (7.3b)
\]

where \( T \) is temperature in degrees centigrade. More details on these coefficient values are provided in the WABASIM report [7.3].

Because \( y_i \) depends on the main variable of the model, the amount of phytoplankton \( x_j \), there is no simple analytic solution to Eq. (7.2), unless we make an additional assumption. One possibility is to assume that detritus achieves a steady state in one time-step of the model. As an alternative it could be assumed that the total live phytoplankton biomass is constant, which makes it possible to solve Eq. (7.2) analytically. In both cases \( y_i \) can be solved from Eq. (7.2), and both solutions are optional in BLOOM II.
Using the (nominal) steady state assumption for recycling, we obtain:

\[
y_i = \frac{\sum (q_i^M x_i^a_i^x_j^*)}{u_i + s}
\]  

(7.4)

Assuming that the total amount of nutrient \( i \) in phytoplankton is nearly constant for a week, Eq. (7.2) may also be solved as an ordinary first order, inhomogeneous differential equation. Because the coefficients in the equations change during the time-step, there is a (usually small) discontinuity in the computed \( y_i \) values at the end of each time-step, when the model starts with new values. This second solution is:

\[
y_i = (y_{i,1} - y_{e,1}) \exp\left\{-(u_i + s)\frac{t}{\text{DELT}_{tt}}\right\} + y_{e,1}
\]  

(7.5)

where

- \( y_{i,1} \) = the present value of \( y_i \),
- \( y_{i,1} \) = the initial value of \( y_i \) at the beginning of a time-step,
- \( y_{e,1} \) = the equilibrium value of \( y_i \), calculated according to Eq. (7.4),
- \( \text{DELT}_{tt} \) = the time-step of the model.

If the amount of detritus were always small compared to nutrients in living phytoplankton, the validity of the steady state assumption would be unimportant, but this is not the case. Usually the size of both nutrient pools are of the same order of magnitude. Therefore it is necessary to discuss how closely \( y_i \) approaches equilibrium.

According to Eq. (7.5), this depends on (1) the rate of approach \((u_i + s)\) multiplied by the time-step, and (2) the difference between equilibrium and initial value of the detritus concentration.

The characteristic time of Eq. (7.4) for the nominal version of the model \((s \text{ is small or completely neglected)}\) is \(1/u_i\) days, which is on the order of a week in summer to a month under winter conditions. Thus the nominal one-week time-step of the model is rather short, at least in winter. Fortunately the difference between the initial and the equilibrium values of the detritus pools \((y_{i,1} - y_{e,1})\) tends to be rather small because all the important processes such as phytoplankton mortality, remineralization, and changes in the total amounts of nutrients are rather smooth functions of time (actually of temperature). Thus errors from these steady state assumptions seem
to be tolerable under ordinary conditions. Indeed, it was shown in the WABASIM report [7.3] that for most natural lakes the results of BLOOM II using Eq. (7.4) are essentially similar to those obtained with the alternative solution Eq. (7.5).

Also notice that both the steady state solution or any kind of dynamic solution depends on the ratio \( q_i^{\times M_j} / (u_i + s) \). At low temperatures both the numerator and the denominator become rather small, hence the detritus pool size is very sensitive to the values of these coefficients; errors in any of these may well have a larger impact on the final results than the type of solution used in the model.

7.3.3. Formulation of Nutrient Constraints

Having obtained two solutions for the detritus equations, we can easily eliminate \( y_i \) from the set of mass balance equations (7.1), which can be rewritten with the phytoplankton concentrations \( x_j \) as the only variables. For brevity, only the steady state solution will be given here, but both are described in Ref. 7.3 (and included in the model). Substituting Eq. (7.4) into Eq. (7.1) gives

\[
\sum_j \left( \frac{u_i + s + q_i^{\times M_j}}{u_i + s} \right) a_{i,j} x_j + e_i = b_i 
\]

(7.6)

This set of equations will be used as constraints by the algorithm of Sec. 7.5 that computes the phytoplankton biomass concentrations.

7.4. ENERGY REQUIREMENTS OF PHYTOPLANKTON SPECIES

The division into autotrophic and heterotrophic organisms is one of the most important ways to classify living creatures into functional units. Photoautotrophic organisms, such as all the well-known terrestrial plants, phytoplankton, and many bacteria, are essentially independent of other creatures for survival because sunlight is their main source of energy. There is no doubt that life on earth depends completely on the photosynthetic \(^{*}\) products of the plants (energy-rich compounds and oxygen), because these are required by all other organisms, whether directly or indirectly (e.g., as fossil fuels).

Plant cells contain a number of light-sensitive pigments such as chlorophyll-a and -b, which can absorb light quanta (photons). In a complicated sequence of photochemical reactions, the energy from the photons is transferred to and stored as readily available chemical energy.
Photosynthesis, which is one of the most complicated physiological processes, varies with circumstances and between species. There are great differences in the preferred light intensities and in the ability of species to adapt to new light or temperature conditions (Fig. 7.1). As shown by Jorgensen [7.4], Van Liere [7.5], and others, many adaptations involve a change in internal chlorophyll levels, which is one of the main reasons why chlorophyll can only be an approximate indicator of phytoplankton biomass, as will be discussed in Sec. 8.2.1.

It is difficult to compute the carbon fixation rates of phytoplankton because the light intensity varies with (1) the seasonal rhythm, (2) the daily rhythm, and (3) the water depth. The last factor is particularly important since light attenuates exponentially with depth, and because many Dutch waters have a high turbidity. Thus, the lowest light intensity at which photosynthesis is still possible (the euphotic depth) is usually found far above the bottom. Meanwhile the light intensities in the surface layers are supersaturating to photosynthesis. An additional complication is the ability in some groups of species to avoid homogeneous mixing (Sec. 7.4.9 and 16.4.2). Superimposed on all variations are weather-induced light fluctuations.

As a final complication, carbon fixation and growth are not always synonymous. Photosynthesis is a prerequisite to, but not a sufficient condition for, growth because all cell materials, not just carbon, must be assimilated before a cell division can occur. Under certain conditions carbon fixation may continue, although actual growth is prohibited by, for example, a nutrient limitation; in that case the surplus amount of fixed carbon must be excreted by the cell.

For simplicity most coefficients in the remainder of this chapter are written without a subscript j, but actually they depend on the species and are treated as such by the model.

7.4.1. The Energy Budget of Phytoplankton

Phytoplankton blooms develop when, during a substantial period, the environmental conditions enable the population to fix more energy than required to compensate for all current losses. Energy gains are determined by the rate of production, which is a function of the solar intensity, the surface reflectance, the attenuation in the water (mixing depth, background extinction, contribution of living and dead phytoplankton to extinction), the day length, the water temperature, perhaps the spectral distribution, or the variations in light intensity.

Energy is lost by several processes: respiration (to some extent the opposite of primary production), mainly for maintenance; mortality (old or unhealthy cells die); grazing by zooplankton or fish; and sedimentation (most phytoplankton species have positive sinking rates).
Fig. 7.1--Photosynthetic efficiency curves of phytoplankton species in the model (data derived from Grote Rug measurements by the Delta Department)
Define the following symbols:

\[ \begin{align*}
\text{Pg} & = \text{the depth and time averaged gross production rate constant per day,} \\
R & = \text{the respiration rate constant per day,} \\
M & = \text{the natural mortality rate constant per day, including all death processes except grazing,} \\
G & = \text{the mortality rate constant per day due to grazing by zooplankton.}
\end{align*} \]

The net effect of these processes may be summarized in the well-known differential equation:

\[ \frac{dx}{dt} = (\text{Pg} - M - R - G) x \quad (7.7) \]

Notice that no functional relations are indicated, which could suggest there are no feedback mechanisms to prevent unlimited growth. Actually the gross production rate \( \text{Pg} \) depends on the light intensity, and therefore on the light attenuation caused by the phytoplankton cells themselves. Thus, as the number of cells grows, the light intensity (and hence the energy available) at any depth is reduced, a phenomenon called self-shading. \( \text{Pg} \) also depends upon nutrient availability and declines when nutrients are scarce. However, when the light intensity is optimal, nutrients are abundant, and all other conditions are favorable, phytoplankton will achieve a maximum gross production rate, which we think of as an innate characteristic of each species of phytoplankton.

Logically, our next section should discuss the maximum gross production rate. Unfortunately, it is much easier to measure the maximum net production rate, which is the maximum gross rate less respiration. It is the net rate we discuss next.

7.4.2. The Maximum Net Production Rate

The maximum gross production rate, \( \text{Pg}_{\text{max}} \), cannot easily be observed directly, since phytoplankton always respire. But under favorable conditions, natural mortality and grazing can be eliminated for all practical purposes, leaving a maximum net production rate, \( \text{P}_{\text{max}} \).

In a classical paper on the relation between temperature and growth of phytoplankton species in the sea, Eppley [7.6] plotted more than a hundred observations on the maximum number of doublings per day (\( \text{U}_{\text{max}} \)) from continuous cultures against \( T \) and then drew an envelope curve through the highest points at each temperature. This curve was described by the following equation:

\[ \text{U}_{\text{max}}(T) = \text{EXP}(0.0639T - 0.16) \quad (7.8a) \]
It is easy to demonstrate that one must multiply the doublings per
day by the logarithm of 2 to obtain the daily growth rate. Thus,
the maximum net growth rate can be expressed as:

\[ P_{\text{max}}(T) = \log_2 \times \exp(0.0639T - 0.16) \quad (7.8b) \]

Since Eq. (7.8a) was derived for single-celled marine phytoplankton,
it could be incorrect for fresh waters, or multicellular species. A
similar temperature dependence has, however, been observed frequently
for several groups of saltwater and freshwater species, for example, by
Foy et al. [7.7], Goldman and Carpenter [7.8], Sakshaug [7.9],
and Harris [7.10], and was also computed for Grote Rug from
measurements by the Environmental Division and the RID.

Large species such as blue-green phytoplankton colonies cannot
achieve such high growth rates as predicted by Eq. (7.8b).
Therefore, following a proposal by Laws [7.11], an additional term
was introduced, depending on the species' volume, which reduces
\( P_{\text{max}}(T) \) of the blue-green species in the model by 0.70.

Defining \( V_j \) = the average volume of species \( j \) in cubic microns and
multiplying several constants, we replace Eq. (7.8b) by:

\[ P_{\text{max}}(T,V) = 1.0729 \times \exp(0.0639T - 0.16) \times V \quad (7.9) \]

As may be verified, Eq. (7.9) is exactly the same as Eq. (7.8b) for a
small species with a volume of 500 \( \mu \)m, and \( P_{\text{max}}(T) \) is reduced to
70 percent for a large blue-green phytoplankton with a volume of
80000 \( \mu \)m.

7.4.3. Respiration

To obtain the maximum gross production rate, we must add the
respiration rate to \( P_{\text{max}}(T) \). There are two different respiratory
processes: photorespiration and dark respiration, of which only the
latter will be discussed here, as, according to Harris [7.10], the
rate of photorespiration of phytoplankton is probably small under
conditions of a bloom. In the process known as dark respiration,
substrates of the previous light period are used by the cell to
synthesize energy-rich compounds and carbon skeletons. Contrary to
what its name suggests, dark respiration proceeds in the light at
approximately the same rate in many species.

Strickland [7.12], Harris [7.10], and many others have shown that
instantaneous dark respiration rates are difficult to measure, and
moreover the usually employed \( ^{14}C \) method to estimate primary
production does not allow respiration to be measured as well. Thus there are comparatively few data on respiration in the literature.

Usually R(T) is described as an exponential function of temperature of the general type:

\[ R(T) = \exp(A \times T - B) \]  

(7.10)

which at 20 deg C varies between 0.05 and 0.20 times the maximum net production rate constant Pnmax(T). But higher respiration rates have been observed frequently for dinoflagellates. The ratio to Pnmax(T) decreases when temperatures decline since R(T) is a stronger function of T than Pnmax(T). According to Harris [7.10], R(T) increases by a factor of 2.3 to 2.9 for every 10 deg C temperature rise. The specific values of A and B were estimated from many references, which are included in Ref. 7.3. Table 7.2 gives typical quantities from which A and B can be calculated.

<table>
<thead>
<tr>
<th>Group of Species</th>
<th>R(20)/Pnmax(20)</th>
<th>Q10</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blue-greens</td>
<td>0.05</td>
<td>2.5</td>
</tr>
<tr>
<td>Diatoms</td>
<td>0.125</td>
<td>2.5</td>
</tr>
<tr>
<td>Flagellates</td>
<td>0.125</td>
<td>2.5</td>
</tr>
<tr>
<td>Greens</td>
<td>0.125</td>
<td>2.5</td>
</tr>
<tr>
<td>Dinoflagellates</td>
<td>0.20</td>
<td>1.9</td>
</tr>
</tbody>
</table>

NOTE: To compute R(T) from Pnmax(T), use Eq. (7.9).

Finally, we can calculate the maximum gross production rate, Pgst(T), to be:

\[ Pgst(T) = Pnmax(T) + R(T) \]  

(7.11)

7.4.4. Natural Mortality

Although respiration and natural mortality are both loss terms, they are physiologically quite distinct. Respiration is mainly (though not entirely) a loss of carbonic substances (energy). But a healthy cell will avoid respiring proteins, for example, unless conditions become extremely unfavorable. Thus respiration will not affect any of the nutrient cycles in the model.
After a number of cell divisions, particularly under unfavorable environmental conditions, any phytoplankter will stop producing new cells and will eventually die. Afterward it disintegrates and releases its cell contents to the surrounding water. A certain fraction will be released in a form that is available to living cells (our present estimate is 50 percent; see Sec. 7.3.2), but the rest will be lost temporarily to the detritus pool and has to be remineralized. Thus mortality has two effects on a phytoplankton population: like respiration, it is a loss of energy, but because the whole cell disintegrates, it is also a loss of cell material (available nutrients).

Unfortunately there are no direct ways to measure the natural mortality rate constant because there is no selective distinction between dead cell materials and other organic substances. The only way to estimate M(T) is to subtract observed net changes in biomass from measured rates of production. Employing this indirect method to Grote Rug data for WABASIM, we concluded that (1) both mortality and production rates are large (up to 1.0 per day) compared to the observed net rates of change in phytoplankton populations (usually less than 0.1 per day), (2) M(T) is exponentially correlated to the water temperature, but (3) exceptionally low values can occur at any temperature.

Annual time series of M could not be related to any variable in a way that enables mortality rates to be computed in agreement with observed numbers. For instance, the frequently assumed negative feedback between M and x (steadily increasing mortality rates as a bloom rises) proved to be incorrect in Grote Rug, where M was usually low during a bloom, but high when there was little phytoplankton.

Therefore an alternative approach was adopted in agreement with the model's purpose to estimate the highest feasible phytoplankton biomass concentrations. Obviously these are obtained if the mortality is set to the minimum value that might reasonably be expected. An equation to compute Mmin(T) was obtained from Grote Rug drawing an envelope curve through the lowest points in a plot of mortality rates against temperature:

\[ M_{\text{min}}(T) = \exp(0.098T - 3.219) \]  \hspace{1cm} (7.12)

Of 204 data points, only 12 (6 percent) fell below this curve, mostly during one particular bloom. Considering present knowledge and the similarity in the estimates of Mmin(T) for blooms of different species, we currently use the same overall estimate for each species. Notice that this assumption does not imply that the actual mortality rates of different species should always be the same, but rather that all could have the same low mortality rate under favorable circumstances.
7.4.5. Primary Production

The rate of primary production is determined by many environmental conditions, which moreover have a different impact on different species. Therefore each mathematical equation will always be a (rather simple) abstraction of reality. In the following the gross production rate constant $P_g$ will be written as the product of various terms, which presumably act independently.

Given sufficient nutrients, $P_g$ is a function of temperature and light, which achieves a single maximum for any given temperature $T$, called $P_g^{\text{max}}(T)$, at an optimal light intensity that we call $I_{\text{opt}}$. If only a small portion ($e_i$) of nutrient $i$ is left available, $P_g$ could decrease by a factor $g(e_i)$, which is equal to or less than one according as $e_i$ is large or small. Similarly, if light energy is scarce, $P_g$ will decrease by a factor of $E(I,T)$, which depends on both temperature $T$ and light intensity $I$: 

$$ I = \text{the total radiation in Joules/m}^2/\text{hr of the photosynthetically active part of the light spectrum.} $$

$P_g$ may be calculated according to:

$$ P_g(I,T,e_i) = E(I,T) \times P_g^{\text{max}}(T) \times g(e_i) \quad (7.13) $$

Various expressions (e.g., the so-called Monod and threshold approaches) have been proposed for $g(e_i)$. But it was argued in Ref. 7.3 that BLOOM II does not require an explicit functional form for $g(e_i)$, but rather:

$$ g(e_i) = 1 \quad \text{for any species limited by energy} \quad (7.14a) $$

$$ g(e_i) < 1 \quad \text{for any species limited by a nutrient} \quad (7.14b) $$

Hence in the case of an energy limitation, nutrients do not affect growth rates; in the case of a nutrient limitation, the bloom will be calculated according to Eq. (7.6) as if the growth rate were to remain sufficiently high for phytoplankton to achieve the nutrient-limited bloom level within a time-step. This approach, of course, is consistent with the purpose of the model.
7.4.6. Production Efficiency and Light

Since the mid-1950s, when techniques to measure photosynthetic rates were greatly improved, many results have been obtained relating production to light intensity. Some of these earlier results, especially those by Ryther [7.13], have become classical and are reproduced by Parsons and Takahashi [7.14] and Bougias [7.15] in their books on aquatic ecology. Usually production is an optimum-type function of light. The production rate increases with increasing intensity I, as long as I is below I_{opt}. For intensities above I_{opt}, the production rate decreases as intensities increase. Thus, I_{opt} is the intensity at which the production rate is maximal.

To compare the curves of different species, we standardize them (i.e., divide by P_{gmax}(T)). The standardized curves, which we call efficiency curves, range from 0.0 to 1.0. It is obvious from Eq. (7.13) that without nutrient limitation this scaling procedure exactly gives us the function E(I,T) for BLOOM II at a particular temperature.

Because several groups of species are included in BLOOM II, an efficiency curve should be specified for each. Preferably these curves would have been gathered under standardized, laboratory conditions, but unfortunately such experiments have not yet been performed for many different species. Currently the curves of Fig. 7.1 are used. These curves were measured by the Environmental Division in Grote Rug for periods (1) when one species was dominant (RID data), (2) the light intensities and nutrient concentrations were similar, and (3) temperature was (approximately) 15°C. If several curves were available for one species, there was usually little variation in the results, indicating good replicability of the measurements.

Given sufficient nutrients, g(e_{i}) = 1.0 for each nutrient i. Hence it follows from Eqs. (7.13) and (7.14a) that:

\[ P_{g}(I,T) = E(I,T) \cdot P_{gmax}(T) \]

and:

\[ E(I,T) = \frac{P_{g}(I,T)}{P_{gmax}(T)} \quad (7.15) \]

To compute the average efficiency in the model, E(I,T) must be integrated over I, as will be shown in Sec. 7.4.7. As, however, P_{gmax}(T) is an exponential function of temperature, E(I,T) is a
function of T, too. Thus we shall find a different value for the average efficiency for each temperature, even if I is constant. This is a great computational disadvantage, which could force us to reintegrate E(I,T) for each temperature.

It was demonstrated in the WABASIM report [7.3] that for physiological reasons E(I,T) and Pmax(T) should have exactly the same temperature dependence and that indeed this was the case in Grote Rug. Thus it could be shown that it is possible to transform E(I,T) to a function of a single variable I', where

\[ E(I') = E(I,T) \] (7.16a)

and:

\[ I' = I \times \exp (-0.0639T + 0.96) \] (7.16b)

Notice that I' = I at T = 15 deg C. (We might call I' an equivalent intensity.) To compute the average efficiency at a given temperature T and light intensity I, the model (1) integrates the standard, 15 deg C efficiency curves and puts the results in a table, (2) transforms the light intensity I to the corresponding 15 deg C value I', and (3) looks up the average efficiency at this transformed intensity level I' (and not I).

7.4.7. Averaging the Production

The light intensity encountered by, and hence the efficiency E(I,T) of, a phytoplankton cell, is not constant but varies with the water depth, turbulence, and time. To account for these variations, the model must compute the average efficiency EAVG in a certain period. Several biological and mathematical complications are involved in the averaging procedures [7.1] and [7.3], which are not discussed here. Introducing a function F and defining the following symbols:

- DL = the average day length in hours,
- I_0 = the surface light intensity in joules/m^2/hr,
- Z_{max} = the maximum mixing depth of the water,
- K = the total extinction of the water per m,
- K_l = the total extinction due to living phytoplankton per m,
- K_d = the total extinction due to detritus per m,
- K_b = the background extinction of the water per m,

EAVG may ultimately be computed as:
\[ DL \ F(K, Z_{\text{max}}, I_0) \]

\[ EAVG = \frac{K \cdot Z_{\text{max}}}{16} \quad (7.17) \]

In this equation,

\[ K = k_1 + k_d + k_0 \quad (7.18) \]

EAVG is proportional to the day length divided by 16 in Eq. (7.17), because growth of many species saturates at a day length of 16 hours. Thus if all other conditions were the same, EAVG would be about twice as low in winter as in summer in the temperate climatic regions. It can be shown that EAVG decreases when the product of the maximum depth and the total extinction increases; thus when a bloom goes up and K rises, EAVG becomes progressively smaller (self-shading).

The results of Eqs. (7.17) and (7.15) could be substituted into the differential equation (7.7) for each species j:

\[ \frac{dx}{dt} = [P_{\text{max}}(T) \cdot EAVG - M_{\text{min}}(T) - R(T) - G] \cdot x \quad (7.19) \]

Using the appropriate expressions for \( P_{\text{max}}(T) \), EAVG, \( M_{\text{min}}(T) \), and \( R(T) \) (remember that \( G = 0 \) had to be assumed), it is now possible to solve Eq. (7.19) numerically. There is no analytical solution, however, because EAVG is a function of K, hence a (complicated) function of x.

7.4.8. Steady State Assumption

Rather than solving Eq. (7.19), BLOOM II assumes a steady state with a nominal time-step of a week. This assumption has been thoroughly discussed for WABASIM [7.3] and will only briefly be summarized here. Rapid fluctuations in phytoplankton populations have been observed frequently, under both laboratory and natural conditions. Usually, however, concentrations change more gradually, which could be caused by a slow approach to some steady state, but also by slowly altering environmental conditions.

To investigate how rapid the approach to equilibrium could be under various conditions, Eq. (7.19) was solved numerically with coefficient values which are typical for various parts of the year and with several (small) initial biomass concentrations. It was concluded that in summer, when usually the worst blooms occur, phytoplankton populations respond rapidly to changes in environmental conditions, although a complete shift in species dominance usually takes more than one week.
In spring, two or three weeks may be required to approach equilibrium; large fluctuations in species composition generally take still longer periods. Under (severe) winter conditions, biomass can only change slowly, implying that equilibrium can only be approached in a week if the initial value is already close to it. Changes in species composition are therefore unlikely.

Considering the purpose of the model (a management model to predict size and composition of objectionable blooms), we concluded that a steady state with a one-week time-step could reasonably be assumed, because the worst blooms occur in summer. But there are obviously periods when this time scale could be too short, particularly if there is a shift in dominance. Whether the model overpredicts during these periods is determined, however, not only by the growth rate, but also by the initial biomass. If the steady states of succeeding periods have about the same value and composition, the model's prediction may still be correct. To investigate (possible) deviations from equilibrium, BLOOM II was extended with alternative options for the time scale, nutrient recycling, and the maximum possible growth rate of individual species, as was already mentioned in Sec. 7.1. A more detailed description of these options can be found in the WABASIM report [7.3].

Under the assumption that each species $j$ is at steady state, obviously

$$\frac{dx}{dt} = 0$$

which can only be true if either:

$$P_g - M_{min}(T) - R(T) - G = 0 \quad (7.20)$$

or:

$$x = 0 \quad (7.21)$$

For each species $j$, either (7.20) or (7.21) should hold. Substituting (7.14a) and (7.14b) for $g(\epsilon_j)$ into (7.13), it follows that

$$P_g \leq E(I,T) \times P_{g_{\text{max}}}(T) \quad (7.22)$$
which after substitution into (7.20) gives:

\[ E(I,T) \geq \frac{M_{\text{min}}(T) + R(T) + G}{P_{\text{gmax}}(T)} \]  

(7.23)

Any species for which (7.23) does not hold has a negative net growth rate and is unable to sustain itself; thus it is excluded from the bloom. The value of \( E(I,T) \) for which (7.23) is an equality is called the minimum efficiency requirement, \( E_{\text{min}}(T) \). This is the lowest possible value of \( E(I,T) \) at which gains and losses are balanced.

Next \( E_{\text{AVG}} \) is set equal to \( E_{\text{min}}(T) \), and then Eq. (7.17) is solved for \( K \). Because of the shape of the efficiency curves, there are usually two roots, one called \( U_{\text{Kmin}} \), the other \( U_{\text{Kmax}} \); the first is the limit where the light intensity becomes too high (photoinhibition), the second where it becomes too low (energy limitation). Since these two roots include only the physiological responses of the model's species to the light regime as if there were no background extinction, the latter must be subtracted from both roots, hence:

\[ K_{\text{min}} = U_{\text{Kmin}} - K_b \]
\[ K_{\text{max}} = U_{\text{Kmax}} - K_b \]

(7.24)

A species can only sustain a positive net growth rate if the total extinction is between its \( K_{\text{min}} \) and \( K_{\text{max}} \) value:

\[ K_{\text{min}} \leq K \leq K_{\text{max}} \]  

(7.25)

Under unfavorable conditions there may be no root for \( K \), implying that the average light intensity is too low (or, conceivably, too high) for production to compensate total losses.

7.4.9. Formulation of Energy Constraints

Besides absorption by background material for which Eq. (7.24) is corrected, the model must also take the extinction by dead phytoplankton particles into account. Defining:

\[ v = \text{the rate constant by which the effect of dead phytoplankton on the extinction diminishes}, \]
\[ K_j = \text{the specific extinction of living and dead material of phytoplankton species j in m}^2/\text{mg dry weight}, \]
and reintroducing subscripts, one may write an analogy to the detritus equations for nutrients of Sec. 7.3.2.:

\[
\frac{dK_d}{dt} = \sum\left(q^{\delta}M_{\text{min}}^{\delta}K_{j}^{\delta}x_{j}\right) - v^{\delta}K_d - s^{\delta}K_d \tag{7.26}
\]

As discussed in the WABASIM report [7.3], the model assumes that (1) \( q = 0.5 \), so that dead phytoplankton particles lose 50 percent of their light-absorbing properties almost instantaneously, and (2) \( v \) can be calculated as an exponential function of temperature:

\[
v = \exp(0.0296T - 1.897) \tag{7.27}
\]

Also since this equation yields values between 0.15 and 0.30 for the normal temperature range, it is reasonable to assume \( K_d \) to be at steady state. Thus:

\[
K_d = \frac{\sum\left(q^{\delta}M_{\text{min}}^{\delta}K_{j}^{\delta}x_{j}\right)}{v + s} \tag{7.28}
\]

Using the temperature dependences for \( M_{\text{min}} \) and \( v \) and a value of 0.5 for \( q \), we can easily verify that \( K_d \) is on the order of 0.20 to 0.40 times the extinction of living phytoplankton \( K_1 \). Thus the computations are less sensitive to errors in \( K_d \) than they are to errors in the steady state detritus pools of the nutrients.

It has proved to be more difficult to model energy by linear constraints than to model nutrients. However, the transformation of light into extinction makes it possible to cast energy and nutrient equations in the same mathematical framework. The energy constraints of species \( j \) follow logically from condition (7.25), substituting the appropriate expressions for living and dead phytoplankton for \( K \):

\[
K_{\text{min}} \leq \sum K_{j}^{\delta}x_{j} + K_d \leq K_{\text{max}} \tag{7.29}
\]
Substitution of (7.28) into (7.29) gives the final energy constraints:

\[
K_{\text{min}} \leq \sum_j \left( \frac{\nu + s + q^x M_{\text{min}}}{K_j x_j} \right) \leq K_{\text{max}} \tag{7.30}
\]

One final remark: not all species are homogeneously mixed over Z_{\text{max}} meters and of the groups of species included in the model, two frequently regulate their vertical position in the water column—blue-green phytoplankton and dinoflagellates. Thus they can absorb photons which would otherwise be absorbed by background material. It was shown for WABASIM that these species can increase their maximum energy-limited biomass by a factor of 1.5 to infinity,\textsuperscript{16} but the empirically determined typical increase is on the order of a factor of 2. Although none of the equations will be shown here, this same advantage has been taken into account in all of the runs for PAWN.

7.5. SOLUTION ALGORITHM

The energy constraints, which were set up in the previous section, permit each species to grow within certain limits of the total extinction coefficient of the water. As shown in Sec. 7.4.8, usually \( K_{\text{min}} \) is negative for each species \( j \) and \( K_{\text{max}} \) is positive for at least some species. Thus the feasible ranges of extinction limits of the different species generally overlap.

In the solution algorithm of BLOOM II, all \( K_{\text{min}} \) and \( K_{\text{max}} \) values are ordered, resulting in \( 2N-1 \) or less extinction intervals, if there are \( N \) species in the model. Below the smallest \( K_{\text{min}} \) value, no species can grow, and of course if \( K \) exceeds the largest \( K_{\text{max}} \) value, no species can grow either. In between these absolute limits, each interval has a different set of species associated with it, which could have positive growth rates. In general most species can grow in the lowest intervals, but as the extinction increases, only a few can maintain a positive net growth rate, as self-shading becomes limiting to an ever larger number of species.

Because the total extinction cannot possibly have two different values at the same time, a separate LP is set up and solved for each interval, always with the same three nutrient constraints, because these should hold regardless of the energy conditions. Thus BLOOM II solves the following LP for each interval \( i \), which is characterized by its extinction limits and its subset of species \( S_i \):
Find: \( x_j \geq 0 \) and \( e_i \geq 0 \) (for subset \( S_1 \)),

Maximizing: \( \sum_j x_j \),

Subject to:

\[
(u_i + s + q_i^{*Mmin}) \sum_j a_{i,j} x_j + e_i = b_i
\]

\( u_i + s \)

And:

\[
(v + s + q_i^{*Mmin}) K_{min,j} \leq \sum_j K_{j} x_j \leq K_{max,j}
\]

\( v + s \)

The LP of each interval \( l \) is solved, and the computed maximum biomass \( B_{max_l} \), the species composition, and limiting factors are recorded. When all LP's for a time-step have been solved, their \( B_{max_l} \) values are compared, and the interval with the highest \( B_{max_l} \) value contains the ultimate solution of BLOOM II; the others may be considered as local maxima. This procedure is repeated for each single time-step.

7.6. BEHAVIOR AND PERFORMANCE

In accordance with its purpose, BLOOM II has been set up in a way that makes overpredictions more likely than underpredictions: it computes maximum equilibria rather than actual biomasses. Thus it is important (1) how often the model overpredicts, and (2) by what magnitude. A simple approach would be to compare the outputs of the model with observed biomass and record how often the former exceeds the latter. However, a 10-percent overprediction has a different significance than a 500-percent overprediction, both of which occur. Recording the frequency without the magnitude of the violations is obviously too simple. In addition, an overprediction is of no significance if the predicted biomass is nonetheless very small.

Therefore we will use the following definition. Any predicted bloom that (1) exceeds the provisional chlorophyll standard and (2) exceeds the observed concentration by more than 50 percent will be considered an overprediction. A 50-percent difference is still regarded acceptable because, as will be pointed out in Sec. 8.2.1, there is an uncertainty of at least a factor of two in the conversion of the model's primary biomass output (in mg dry weight per m\(^3\)) to
chlorophyll. Hence even at a 50-percent deviation between predicted and observed chlorophyll, the weight prediction of the model could still be correct.

NOTES

1. Zero nutrient transport is assumed only in BLOOM II, but several transport mechanisms are included in CHARON and OXIMOD.
2. If the changes in $x_j$ within a week were considered too large, the algorithm could be extended with an additional corrector step, using the average values of the $x_j$'s of two time-steps to recalculate the values of $y_i$. These new values of $y_i$ could then be used in the linear program to recompute the $x_j$'s, and the $y_i$'s could be recalculated once more, etc.
3. The model's results for those Grote Rug reservoirs in which P is chemically precipitated show greater differences from observations because the time-dependent simulation of detritus is strongly affected by the artificially induced variations in the sedimentation rates.
4. Photosynthesis is here defined as photochemical carbon fixation.
5. Growth is here defined as increase in biomass (dry weight); it requires a balanced uptake of all essential elements in addition to carbon and is usually followed by a cell division.
6. Short periods with high intensities and long periods with moderate intensities might have a different impact, even if the total number of quanta received is the same.
7. Let:
   \[ d = \text{doublings per day}, \]
   \[ r = \text{daily growth rate}, \]
   \[ C(0) = \text{initial phytoplankton concentration}, \]
   \[ C(t) = \text{phytoplankton concentration after } t \text{ days}. \]

Then, by definition,

\[ C(t) = C(0) \times 2^{d \times t} = C(0) \times \exp(r \times t) \]

Taking the natural logarithm throughout, we conclude that:

\[ r = d \times \log(2) \]

8. Currently a WABASIM-sponsored research project at the Microbiological Laboratory of the Amsterdam University is investigating the growth rates of six phytoplankton species at different temperatures and light intensities.
9. In most eutrophic waters, Kmin of any species $j$ is smaller than the background extinction; hence the average light intensity is too low for photo-inhibition to be of major importance.
10. Occasionally these species can only grow if they are inhomogeneously mixed; if they were mixed over the total depth, the net growth rate would be negative.
REFERENCES


Chapter 8
DATA AQUISITION FOR BLOOM II

8.1. INPUTS TO THE MODEL

BLOOM II has two kinds of inputs: universal and lake-specific. The universal inputs were initially determined from (many) literature sources and Grotse Rug observations but were sometimes modified while calibrating the model. They were then used without change for all PAWN lakes. The lake-specific inputs were directly or sometimes indirectly determined from measurements on individual lakes.

8.1.1. Universal Inputs

Most of the universal inputs have already been discussed here or are discussed in the WABASIM report [8.1]. Summarizing, BLOOM II requires the following universal inputs:

1. The remineralization rates \( u_i \) of the nutrients as a function of temperature.
2. The fraction of dead phytoplankton (q), which has to be remineralized.
3. Minimum stoichiometric coefficients \( a_{i,j} \) for the nutrients.
4. The ratio of carbon to chlorophyll (C/CHL) for each species \( j \).
5. The disappearance rate of dead chlorophyll (v) as a function of temperature.
6. The specific extinction coefficients \( K_j \) for each species \( j \).
7. The average cell volume \( V_j \) of an individual or colony of species \( j \).
8. The ratio of respiration to maximum gross production and the temperature dependence of respiration.
9. The production curve E(I,T) of each species \( j \).
10. A buoyancy regulation factor \( KRF_j \), explained in the WABASIM report.
11. The average day length.
12. Several grazing coefficients (if zooplankton data are available).
13. The nutrient contents of zooplankton (if zooplankton data are available).
Values for some of the more important universal inputs appear in Tables 8.1 and 8.2.

Table 8.1

<table>
<thead>
<tr>
<th>Species</th>
<th>Average Volume</th>
<th>Respiration Coefficients</th>
<th>A</th>
<th>B</th>
</tr>
</thead>
<tbody>
<tr>
<td>Asterionella</td>
<td>500</td>
<td>0.0916</td>
<td>3.0275</td>
<td></td>
</tr>
<tr>
<td>Centric diatoms</td>
<td>500</td>
<td>0.0916</td>
<td>3.0275</td>
<td></td>
</tr>
<tr>
<td>Cryptomonas</td>
<td>500</td>
<td>0.0916</td>
<td>3.0275</td>
<td></td>
</tr>
<tr>
<td>Volvox</td>
<td>500</td>
<td>0.0916</td>
<td>3.0275</td>
<td></td>
</tr>
<tr>
<td>Scenedesmus</td>
<td>1000</td>
<td>0.0916</td>
<td>3.0751</td>
<td></td>
</tr>
<tr>
<td>Ceratium</td>
<td>20000</td>
<td>0.0639</td>
<td>2.1721</td>
<td></td>
</tr>
<tr>
<td>Anabaena</td>
<td>80000</td>
<td>0.0916</td>
<td>4.3827</td>
<td></td>
</tr>
<tr>
<td>Aphanizomenon</td>
<td>80000</td>
<td>0.0916</td>
<td>4.3827</td>
<td></td>
</tr>
<tr>
<td>Microcystis</td>
<td>80000</td>
<td>0.0916</td>
<td>4.3827</td>
<td></td>
</tr>
<tr>
<td>Oscillatoria</td>
<td>80000</td>
<td>0.0916</td>
<td>4.3827</td>
<td></td>
</tr>
</tbody>
</table>

Notice that occasionally the same volume and respiration coefficients have been assigned to different species in one group to prevent small differences from determining dominance within a group.

Table 8.2

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>P</th>
<th>Si</th>
<th>K_j</th>
<th>C/CHL</th>
<th>DRY/C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Asterionella</td>
<td>0.024</td>
<td>0.0032</td>
<td>0.22</td>
<td>7.00D-05</td>
<td>25</td>
<td>3.0</td>
</tr>
<tr>
<td>Centric diatoms</td>
<td>0.040</td>
<td>0.0046</td>
<td>0.13</td>
<td>7.00D-05</td>
<td>25</td>
<td>3.0</td>
</tr>
<tr>
<td>Cryptomonas</td>
<td>0.072</td>
<td>0.0046</td>
<td>0.0007</td>
<td>5.21D-05</td>
<td>30</td>
<td>2.3</td>
</tr>
<tr>
<td>Volvox</td>
<td>0.076</td>
<td>0.0070</td>
<td>0.0007</td>
<td>3.86D-05</td>
<td>30</td>
<td>2.3</td>
</tr>
<tr>
<td>Scenedesmus</td>
<td>0.058</td>
<td>0.0052</td>
<td>0.0007</td>
<td>5.21D-05</td>
<td>30</td>
<td>2.3</td>
</tr>
<tr>
<td>Ceratium</td>
<td>0.064</td>
<td>0.0046</td>
<td>0.0007</td>
<td>6.00D-05</td>
<td>30</td>
<td>2.3</td>
</tr>
<tr>
<td>Anabaena</td>
<td>0.070</td>
<td>0.0057</td>
<td>0.0007</td>
<td>2.66D-04</td>
<td>30</td>
<td>2.5</td>
</tr>
<tr>
<td>Aphanizomenon</td>
<td>0.068</td>
<td>0.0043</td>
<td>0.0007</td>
<td>2.35D-04</td>
<td>30</td>
<td>2.5</td>
</tr>
<tr>
<td>Microcystis</td>
<td>0.053</td>
<td>0.0057</td>
<td>0.0007</td>
<td>9.60D-05</td>
<td>40</td>
<td>2.5</td>
</tr>
<tr>
<td>Oscillatoria</td>
<td>0.063</td>
<td>0.0046</td>
<td>0.0007</td>
<td>1.90D-04</td>
<td>40</td>
<td>2.5</td>
</tr>
</tbody>
</table>

As with volume and respiration, some stoichiometric coefficients have been considered equal (e.g., for P), since observed data for several species were overlapping. Thus other, more obvious differences determine species dominance.
8.1.2. Lake-Specific Inputs

Contrary to the universal inputs, the lake-specific inputs vary from one lake to another or from one year to the next. These inputs are:

1. The average weekly water temperature.
2. Weekly concentrations of total available N, P, and Si.
4. The background extinction $K_b$ of the water.
5. The mixing depth $Z_{max}$.
6. The flushing rate constant or residence time.
7. Weekly zooplankton concentrations.
8. An adjustment in the specific ratio of carbon to chlorophyll (only in a few lakes).

8.2. DATA AVAILABILITY FOR PAWN LAKES

Frequently data for some of these variables will be missing for some or all time periods. How significant this is to the model's performance depends on what data are missing and how frequently. Obviously, it is impossible to run BLOOM II without nutrient concentrations, but the lack of zooplankton data presumably does not invalidate the results for the eutrophic PAWN lakes.

For no lake are complete weekly time series of all input variables available. The measurement frequency is two weeks in some of the most eutrophic lakes, e.g., Veluwemeer and Wolderwijd, and longer in some other lakes. The time series were completed by linear interpolation, which seems reasonable considering the smooth annual trends of most variables in most places. For waters under national jurisdiction, where data are usually collected at several spots in each lake, average values have been used in the computations.

Because dissolved rather than total silicon is measured, the maximum observed during the first weeks of the year is used for the entire period in which diatom blooms are usually reported (week 1 through 12). For the rest of the year it is assumed that there are no diatoms, and thus dissolved silicon is considered all that is available. By this procedure the total available amount of Si is more likely to be over- than underestimated.

In some lakes silicon was not measured at all, in which case we have used either concentrations from similar lakes or a reasonable annual time series with high initial values, low values after week 12, and a slow increase during the rest of the year. Regardless of the assumed silicon concentration, however, BLOOM II infrequently predicts diatoms after the first quarter of the year (see also Sec. 16.3.1). In the next sections only some general problems will be discussed, but more details for individual lakes are given in Sec. 8.2.3.
8.2.1. Chlorophyll to Biomass Ratio

The carbon to chlorophyll ratio (C/CHL) of a species is not used in the actual computations of the model, but is necessary to convert its dry weight output to chlorophyll, the only commonly measured biomass indicator. Because chlorophyll to biomass ratios may vary considerably, we would prefer not to make any conversion at all, but rather compare BLOOM II's output directly to some observed number such as planktonic dry weight. Unfortunately, it is impossible in routine measurement programs to separate C or dry weight in living phytoplankton from all other fractions.

The C/CHL<sub>j</sub> ratio is not only a function of the species, but also of several abiotic conditions such as temperature, (average) light intensity, and variations in light intensity. For example, a species is more likely to contain a high amount of light-absorbing chlorophyll when it is energy limited than under nutrient-limited conditions. Thus a species-dependent variation in time and place may be expected, and the question arises whether it is at all possible to say anything about C/CHL<sub>j</sub> ratios, other than that they vary over a wide range. Indeed literature values from 20 to 90, sometimes even much higher, have been reported frequently, for example, by Strickland [8.2], Eppley et al. [8.3], and Banse [8.4].

Initial estimates for the C/CHL<sub>j</sub> ratios were obtained from linear regressions between chlorophyll and several phytoplankton-related variables measured in Grote Rug, such as total dry weight, ash free dry weight, particulate organic carbon, and particulate nitrogen and phosphorus. The reader should consult the previous references for a discussion of the methodological problems involved in these estimates. The estimates seem reasonable, because (1) most of the correlations are highly significant, (2) the differences between the various ratios are within the range of expected values, and (3) they are consistent through the years.

In most of the PAWN lakes the average C/CHL ratios agreed well with these Grote Rug estimates. The differences between individual lakes could usually be related to observed differences in species dominance. Hence in the most recent version of BLOOM II, the same specific conversions of dry weight to chlorophyll are used in all but two cases, the Westeinder and the Wolderwijd. Previously, we made an adjustment in some other PAWN lakes as well, but after recent improvements in the predicted composition of species, adjustment was no longer necessary. The average C/CHL ratio of 60 to 65 in the two exceptional lakes is beyond the nominal range of any of BLOOM II's species, however, and thus a lake-specific adjustment for these cases is inevitable.

The model's dry weight computation will always be converted to chlorophyll. However, in comparing blooms under alternative conditions, one should use dry weight rather than chlorophyll, because:
• The conversion depends on the dominant species, thus any change, or the absence of any change in chlorophyll levels, may be due to a shift in composition, rather than a change in dry weight.
• If the average light conditions change drastically, some or all species may adjust their chlorophyll to biomass ratios; thus it is uncertain whether the present conversions still hold in future situations.

8.2.2. The Background Extinction

According to Eq. (7.18), three different components contribute to the total extinction: living phytoplankton, dead phytoplankton, and background material. The background extinction $K_b$ is the value the extinction of the water would have if no phytoplankton were present. In eutrophic lakes, the contribution of living and dead phytoplankton particles is almost always significant, and hence there are no direct measurements of $K_b$. Therefore $K_b$ is usually estimated as the intercept of a linear regression equation of extinction against chlorophyll:

$$K = K_c \text{ CHL} + K_b$$  \hspace{1cm} (8.1)

hence,

$$K_b = K - K_c \text{ CHL}$$  \hspace{1cm} (8.2)

where $K_c$ is the average phytoplankton extinction in $m^2/mg$ chlorophyll, CHL = the chlorophyll concentration in $mg/m^3$.

Unfortunately, instead of $K$, only the Secchi disc measurements, which are inversely proportional to $K$, are available for the PAWN lakes:

$$K = K_{SC}/S$$  \hspace{1cm} (8.3)

where $S$ = the depth of the Secchi disc measurement in m,
$K_{SC}$ = a conversion constant between extinction and Secchi disc.

Thus $K_b$ can be computed as:
\[ K_b = \frac{KSC}{S} - K_c \text{ CHL} \]  

(8.4)

Literature values for KSC range from 1.4 to 3.0, so that using Secchi discs introduces an additional uncertainty in the estimation of \( K_b \) of about a factor of 2. In Grote Rug, where both \( S \) and \( K \) are measured regularly, KSC is usually on the order of 3.0, and this value was also adopted for PAWN because (1) many of the considered lakes have similar bottoms and receive similar (Rijn) water, and (2) smaller values of KSC imply that \( K_c \) should be unrealistically small compared with literature data.

In most PAWN lakes, \( K_b \) could be estimated by the procedure described in the foregoing, but in some lakes correlation coefficients were not significant at a 5-percent level, because of insufficient or scattered data. For these lakes yearly average values of CHL, \( S \), and \( K_c \) were fed into Eq. (8.4) to estimate \( K_b \).

In the Markermeer and IJmeer, Secchi disc measurements vary strongly during a year, even in periods with little or no phytoplankton. Thus it seems unjustifiable to use a single value for \( K_b \). These variations are probably due to stirring of bottom particles and could be correlated with the average wind speed. For these two lakes, blooms were computed both with \( K_b \) as a constant and with \( K_b \) as a weekly variable, computed from the weekly average wind speed. The values of \( K_b \) used in the computations are shown in Table 8.3.

### 8.2.3. Summary of Lake-Specific Inputs

In the previous sections we have discussed the more general problems involved in data acquisition for the PAWN lakes. In many of them, specific problems had to be solved or at least recognized. These will be indicated in Table 8.3, which also shows some important nominal input values.

Solar radiation data were measured by the KNMI and converted from total radiation to photosynthetic active radiation (400-700 nanometers) using a multiplication factor of 0.45.

In the Westeinder, data were only measured at one, atypical location (in a harbor for yachts) during the first 21 weeks of the year. The high observed nitrogen concentrations were later corrected using a regression equation that was established between observed nitrogen at this location and at a more representative location, which was only sampled the second part of the year. As a result, nitrogen concentrations in the first 21 weeks were estimated as approximately 0.44 times the observed levels in the harbor.
Table 8.3
YEARY LAKE-SPECIFIC INPUTS

<table>
<thead>
<tr>
<th>Case</th>
<th>Source</th>
<th>$K_b$</th>
<th>Depth (m)</th>
<th>Nom? (a)</th>
<th>Missing Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>IJsselmeer</td>
<td>1</td>
<td>1.75</td>
<td>4.5</td>
<td>Yes</td>
<td>S: incomplete</td>
</tr>
<tr>
<td>Markermeer</td>
<td>1</td>
<td>4.0</td>
<td>3.3</td>
<td>Yes</td>
<td>Si</td>
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<tr>
<td>Veluwemeer</td>
<td>1</td>
<td>4.5</td>
<td>1.2</td>
<td>Yes</td>
<td>None</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>1</td>
<td>2.5</td>
<td>1.5</td>
<td>No</td>
<td>None</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>1</td>
<td>5.5</td>
<td>1.7</td>
<td>Yes</td>
<td>Si</td>
</tr>
<tr>
<td>IJmeer</td>
<td>1</td>
<td>2.5</td>
<td>2.0</td>
<td>Yes</td>
<td>Si</td>
</tr>
<tr>
<td>Slotermeer</td>
<td>2</td>
<td>6.5</td>
<td>1.2</td>
<td>Yes</td>
<td>Si; T (b); S: incomplete</td>
</tr>
<tr>
<td>Westeinder</td>
<td>3</td>
<td>4.0</td>
<td>2.5</td>
<td>No</td>
<td>Si</td>
</tr>
<tr>
<td>Haringvliet</td>
<td>1</td>
<td>1.2</td>
<td>6.0</td>
<td>Yes</td>
<td>CHL: weeks 1-18</td>
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<tr>
<td>Stuwpand Lith</td>
<td>1</td>
<td>2.25</td>
<td>5.0</td>
<td>Yes</td>
<td>S (c)</td>
</tr>
<tr>
<td>Zoommeer</td>
<td>1</td>
<td>1.2</td>
<td>2.5</td>
<td>Yes</td>
<td>All (d)</td>
</tr>
</tbody>
</table>

SOURCES: 1 = Anon., Kwaliteitsonderzoek in de Rijkswateren, Rijkswaterstaat: RIZA & RID, Kwartaal t/m 4, 1976, usually taken from the WAKWAL database.

(a) The column for C/CHL indicates whether the nominal specific values for each species (Table 8.2) are used (Yes) or lake-specific adjustments are made (No).
(b) Temperature assumed to be the same as in Veluwemeer.
(c) Data used from Keizersveer.
(d) Presently nonexistent lake; all input except depth taken from Haringvliet.

REFERENCES

Chapter 9

CALIBRATION AND VALIDATION

9.1. NOMINAL SIMULATIONS WITH BLOOM II

In 1979 a fairly well calibrated version of BLOOM II was established for WABASIM and applied both to describe the situation existing in each of the PAWN lakes in 1976 (the nominal situation) and to simulate management tactics in the PAWN lakes. Afterward when new Grote Rug and literature data became available, the model was recalibrated and documented in the WABASIM report [9.1]. With this latest version the agreement between the computed and observed blooms improved in some of the lakes considered for PAWN. Therefore all the nominal results for this report have been recomputed and are presented in this chapter.

It was not necessary, however, to recompute the impacts of all eutrophication control tactics, because the rates of change are generally of greater importance than the absolute biomass amounts. Thus all major conclusions would have been essentially similar, if the most recent version of the model had been used.

No universal inputs were changed in an attempt to improve the goodness of fit for the lakes with the worst agreement between observations and computations. Thus the nominal results are in fact validations instead of calibrations.

9.2. MODEL OUTPUTS

Because BLOOM II is a multispecies model with several potential limiting factors, a great deal of output (up to 5000 lines of detailed information) is potentially available, although the output may be limited to only two summary tables if desired. The following outputs are always or optionally produced by the model for each time-step and sometimes for each extinction interval:

1. Concentration of each species in mg dry weight per m³.
2. Total concentration of all species in mg dry weight per m³.
3. Total concentration of all species in mg chlorophyll per m³.
4. Concentration of "planktonic" and "rest" nutrients² in mg per m³.
5. The total extinction per m.
6. Production, mortality, grazing, respiration, and flushing rates of each species in mg C and mg O₂ per m³ per day.
7. Diurnal distribution of the total production and respiration rates in mg O$_2$ per m$^3$ per hour.

For the nominal runs shown here, only the most essential outputs of BLOOM II are shown in the figures accompanying this chapter, namely total chlorophyll and limiting factors. Chlorophyll predictions are directly compared to observations, but this is impossible for limiting factors, because in the measurements there is no weekly indication of what is limiting. A low dissolved nutrient concentration is a necessary but insufficient condition to prove that it is limiting, and for energy there is usually only indirect evidence; if all dissolved nutrients seem sufficient, energy is the only possible limitation left.

The outputs shown in each figure are:

1. Predicted (histograms) and observed (drawn lines) total biomass in mg chlorophyll per m$^3$.
2. Bloom limiting factors, abbreviated as:
   \[\text{Pho} = \text{phosphorus}, \text{Sil} = \text{silicon}, \text{Nit} = \text{nitrogen},\]
   \[\text{Energ} = \text{energy}, \text{and Gro} = \text{growth}.\]
   The last factor can be limiting only if the optional constraints on growth rates are added to the nominal set of constraints (Sec. 7.1), which in this study was only the case in a run for the Markermeer. A star is plotted for each factor that is limiting to the bloom in a time period.

Because the dry weight to chlorophyll conversion of BLOOM II depends on the dominant species, there are sometimes variations in predicted chlorophyll which correspond to a change in composition rather than a change in predicted dry weight. For instance, the three extremely high values for the Wolderwijd after week 40 in 1975 are caused by an erroneous but temporary shift from blue-greens to dinoflagellates; the variation in dry weight over this period is much smaller.

9.3. SOME RESULTS FOR GROTE RUG

Calibration of BLOOM II with Grote Rug data is thoroughly discussed in the WABASTM report. Here only two typical cases are reproduced: Ring 2, 1977 (Fig. 9.1), and Ring 3, 1976 (Fig. 9.2). In the first case the observed mortality rate constants were used as a forcing function to eliminate the uncertainty of using a minimum estimate of mortality, Mmin(T). As may be noted, BLOOM II reproduces both the observed total biomass and the dominant species remarkably well. Thus provided with a good estimator for mortality, BLOOM II can make accurate predictions not only for periods when peak phytoplankton levels are observed, but also for periods without blooms, in accordance with its purpose.
Fig. 9.1--Grote Rug, Ring 2, 1977: Predicted and observed chlorophyll in mg/m³; calibration run with measured mortality rates
Fig. 9.2--Grote Rug, Ring 3, 1976: Predicted and observed chlorophyll in mg/m³; calibration run.
For Ring 3, 1976, the mortality rate constants were calculated according to Eq. (7.9). The model's results are almost as good as in the previous case with observed mortality rates during blooms. But the predicted biomasses in the first half of the summer are higher than observed, although usually below the provisional phytoplankton biomass standard (100 mg chlorophyll per m³) of an objectionable bloom. As in the previous case, species dominance is well reproduced by the model.

9.4. VALIDATIONS (NOMINAL RESULTS) FOR PAWN LAKES

9.4.1. Lake-Specific Results

9.4.1.1. IJsselmeer. In the IJsselmeer (Fig. 9.3), predicted and observed total chlorophyll match well, although the model underpredicts observed levels in the second half of the year. This could be because the net production is underestimated, or because the background extinction or the C/CHL ratio were actually lower than assumed. Most of the year energy is the only limitation, but in the second quarter phosphorus is limiting in addition. The predicted dominance of the blue-green species Microcystis and Oscillatoria agrees reasonably well with the 1976 observations (Table 9.1), but notice that in other years various species of green phytoplankton, e.g., Scenedesmus, have dominated the phytoplankton composition in this lake.

| Table 9.1 |
| SUMMARIZED RESULTS FOR LIMITING FACTORS AND SPECIES COMPOSITION | AS OBSERVED AND PREDICTED IN THE NOMINAL RUNS OF BLOOM II |

<table>
<thead>
<tr>
<th>Lake</th>
<th>Limitation</th>
<th>Composition</th>
</tr>
</thead>
<tbody>
<tr>
<td>IJsselmeer</td>
<td>E; P</td>
<td>B4, B3</td>
</tr>
<tr>
<td>Markermeer</td>
<td>E; N</td>
<td>G1; B3, D</td>
</tr>
<tr>
<td>Veluwe</td>
<td>N, E; P, E</td>
<td>B3; B4</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>N, E; P, N</td>
<td>B3; B3, D</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>E, N</td>
<td>B3; B4, B2</td>
</tr>
<tr>
<td>IJmeer</td>
<td>P; N</td>
<td>B3; B4, B2</td>
</tr>
<tr>
<td>Slotermee</td>
<td>P; N</td>
<td>B3; B4, B2</td>
</tr>
<tr>
<td>Westeinder</td>
<td>N</td>
<td>B3; B3</td>
</tr>
<tr>
<td>Haringvliet</td>
<td>E (?)</td>
<td>B3; B4</td>
</tr>
<tr>
<td>Stuwpond</td>
<td>E (?)</td>
<td>B3; B4</td>
</tr>
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</table>

<table>
<thead>
<tr>
<th>LIMITATION</th>
<th>COMPOSITION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Observed</td>
<td>Predicted</td>
</tr>
<tr>
<td>Observed</td>
<td>Predicted</td>
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<td>B3; B4</td>
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</tr>
<tr>
<td>B3; B4; D</td>
<td>B3; B4; D</td>
</tr>
</tbody>
</table>


NOTES: Abbreviations for limitations: E = energy, N = nitrogen, P = phosphor and Si = silicon. Abbreviations for compositions: D = diatoms, F = flagellates, I = dinoflagellates, G1 = green phytoplankton species Scenedesmus; B = blue-green species: B2 = Aphanizomenon, B3 = Microcystis, B4 = Oscillatoria.
A question mark indicates uncertain or unknown; limitations and species are given in descending order of importance, separated by a comma or semicolon to indicate that the difference is small or considerable.

9.4.1.2. Markermeer. The Markermeer (Figs. 9.4 and 9.5) was connected to the IJsselmeer in 1976; thus these two lakes have many similar physical and chemical characteristics. The most obvious difference is a much greater average value for the background extinction in the Markermeer, which reduces the predicted blooms considerably below the predicted IJsselmeer levels in all but the second quarter of the year. In this period, when N becomes the main limitation, BLOOM II clearly overpredicts, which was never the case in the IJsselmeer.

The reason for these differences could be that the clay bottom of the Markermeer is much more susceptible to stirring by wind than the relatively sandy bottom of the IJsselmeer, which has two important consequences:

1. The average value and the standard deviation of the background extinction are greater.
2. Organic, slowly degrading particles may become dissolved, while nutrients (P in particular) can easily become adsorbed to clay particles and hence unavailable to phytoplankton. But these nutrient fractions are included in the measurements, and BLOOM II considers them available.

Because CHARON had not been developed far enough, it was impossible to determine the unavailable nutrient fraction. As mentioned in Sec. 8.2., the background extinction was correlated with the weekly average wind speed, which fluctuates rather strongly. Thus the results of a second run are shown in Fig. 9.5, in which the background extinction is included as a weekly variable and growth rate constraints are added (Sec. 7.1) to prevent unlimited growth if the background extinction suddenly drops. Obviously (1) the degree of overprediction is reduced, and (2) the computed biomass variations increase considerably in some periods, indicating that the basic steady state assumption of the model might well be violated frequently.

9.4.1.3. Veluwemeer. The Veluwemeer (Figs. 9.6 and 9.7) is obviously the most eutrophic lake considered in this study: phytoplankton concentrations of 500 mg/m³ are not just predicted, but also observed in 1975. BLOOM II tends to predict chlorophyll about 100 mg/m³ too high in some parts of the year (the third quarter), but considering the magnitude of the blooms and the uncertainty in several coefficients such as the dry weight to chlorophyll conversion, it may be concluded that the model computes total biomass reasonably well.
Fig. 9.4--Markermeer, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
Fig. 9.5--Markermeer, 1976: Predicted and observed chlorophyll in mg/m³; weekly background extinctions from wind speeds; additional growth constraints
Fig. 9.7--Veluwemeer, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
In both 1975 and 1976, the limiting factors vary in a complex way: energy, P, and Si are most important during the first quarter; N (sometimes with P) are most important during second and third quarters; and energy, P, and N limit blooms of the fourth quarter. Sometimes there is one limitation, but frequently two, or even more than two, factors are limiting at the same time.

Microcystis is by far the most important species in the computations. Oscillatoria, which actually dominates in this lake, comes in second place because in the model it cannot outcompete Microcystis in summer, when N is limiting. ²

9.4.1.4. Wolderwijd. The Wolderwijd (Figs. 9.8 and 9.9) is similar to the Veluwemeer in some respects, e.g., depth, but there are some important differences. The background extinction is considerably lower (Table 8.3) and the nutrient concentrations may not achieve such extreme values as in the Veluwemeer, but they still make massive blooms possible. Because both the background extinction and the extinction by living phytoplankton are lower, the average light intensity is higher, which is the most likely explanation for the exceptional biomass to chlorophyll ratio in the Wolderwijd. As mentioned in Sec. 8.2.1, approximately 1.5 times the nominal C/CHL ratio has been used for each species in this lake.

The overall agreement between computed and observed total chlorophyll is excellent both in 1975 and 1976. The extremely high peaks predicted after week 40 in 1975 and in the middle of 1976 are not due to a major biomass increase, but to a temporary shift in species dominance. Also notice that BLOOM II computes total chlorophyll equally well in both years, although the observed concentrations in 1976 are considerably lower than in the previous year.

As in the Veluwemeer the limiting factors vary seasonally, but in the Wolderwijd P is at least as important as N, which only becomes the main limitation during the third quarter in both considered years. Rather than computing the observed dominance of a single species, Oscillatoria, BLOOM II computes a complicated seasonal succession of many, particularly blue-green, species (Table 9.1), some of which coexist when more than one factor is limiting to the bloom.

9.4.1.5. Gooi/Eemmeer. In the Gooimeer and Eemmeer (Fig. 9.10), nutrient concentrations differ considerably from those in the Veluwemeer or Wolderwijd. The average phosphorus concentration is much higher, but the opposite holds for nitrogen. Thus, as is observed (Table 9.1), N is by far the most important limiting factor in the computations.

Total biomass increases sharply at the beginning of the year and gradually declines during the rest of it, both in the computations and observations. There is, however, a consistent overprediction of 100 to 200 mg chlorophyll per m³, except during the last quarter. Because N is limiting almost all year, the most likely explanation is
Fig. 9.8—Wolderwijd, 1975: Predicted and observed chlorophyll in mg/m³; nominal run
Fig. 9.9--Wolderwijd, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
Fig. 9.10--Gooimeer, 1976: Predicted and observed chlorophyll in mg/m$^3$; nominal run
an overestimation of the available amount of nitrogen. In other years than 1976, (much) higher biomass levels have been observed and for these the model perhaps would not overpredict. The computed dominance of the species Microcystis is incorrect (Table 9.1), although at least the dominant group is correct.

9.4.1.6. IJmeer. In the IJmeer (Figs. 9.11 and 9.12), nutrient concentrations follow a trend similar to that of the Gooimeer, but the average values are lower. BLOOM II computes phytoplankton concentrations of 200 to 300 mg chlorophyll per m$^3$ in spring and values of 100 to 150 during the rest of the year. Because observed levels exceed 100 mg chlorophyll per m$^3$ only once, the model overestimates bloom levels.

As for the Markermeer, the most plausible explanation is an overestimation of the available amount of the main limitation (in this case N) because (1) the significant correlation between the Secchi disc and the average wind speed might be due to (temporary) release of particulate, unavailable nutrients from the bottom, and (2) much of the nutrient load to the IJmeer is sewage, which usually contains a high unavailable fraction. Using the background extinction as a weekly variable (Fig. 9.12) has little impact on the model predictions, because nutrients remain more important limitations than energy.

9.4.1.7. Slotermeer. The Slotermeer (Fig. 9.13) is another example of a lake in which, according to BLOOM II, a higher than observed biomass should have been possible in 1976. The actual maximum for the year, however, was of the same order of magnitude as the model's predictions for that period. In agreement with the observations, P is indicated as the most important, though not the only, limiting factor.

It must be stressed that (1) many input data were missing (Table 8.3) and that available data were measured infrequently at only one sampling point in the lake and (2) the Slotermeer is a peat lake and thus may contain a considerable particulate, unavailable nutrient fraction. How many nonphytoplankton particles are suspended is illustrated by the extremely high value for the background extinction (Table 8.3).

9.4.1.8. Westeinder. In the Westeinder (Fig. 9.14), predicted and observed chlorophyll concentrations are comparatively low. Only in the first quarter of the year is a bloom computed that slightly exceeds the provisional biomass standard, but the rest of the year predicted chlorophyll concentrations remain below 80 mg/m$^3$. Actual biomasses were exceptionally low in the first half of the year, but during the second half, predicted and observed levels match extremely well. N is the only important observed and predicted limitation.

9.4.1.9. Haringvliet. Actually the Haringvliet (Fig. 9.15) is no lake, but when water is in short supply, the sluices to the North Sea are closed and the average retention time increases from weeks to
Fig. 9.12--IJmeer, 1976: Predicted and observed chlorophyll in mg/m³; weekly background extinctions from wind speeds
Fig. 9.13--Slotermeer, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
Fig. 9.14--Westeinderplasessen, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
Fig. 9.15--Haringvliet, 1976: Predicted and observed chlorophyll in mg/m³; nominal run
months. This was the case during a major part of the dry summer of 1976. To avoid violations of its steady state assumptions, BLOOM II can only consider situations in which the retention times are on the order of a few weeks (in summer) or more than a month (in winter). As may be expected in a deep lake with comparatively high nutrient loadings, energy is the only limiting factor. Thus phytoplankton concentrations must decrease as the retention time becomes shorter, because flushing acts as an increased loss rate. Living phytoplankton are flushed to the North Sea, while water with little or no phytoplankton is coming in from the Rijn. Because usually flushing will not be zero, an average retention time of 20 days was considered as the nominal case.

BLOOM II computes a high bloom (about 180 mg chlorophyll per m$^3$) in the second quarter of the year, when unfortunately chlorophyll was not yet measured. The first observation (around week 20) is about as high as the model's prediction. Phytoplankton concentrations (predicted and observed) decline the rest of the year except for a short period after week 30. As usual, predicted chlorophyll tends to be somewhat higher than observed, but this is of minor importance, since predicted levels are below the provisional standard. Many weeks in winter no species can maintain a positive net growth rate, because the energy limitation is too severe.

9.4.1.10. Stuwand Lith. The Stuwand Lith (Fig. 9.16) is another special case to which BLOOM II can be applied only if the stuw (weir) is closed during a sufficiently long period. Assuming a retention time of 20 days, there is an excellent agreement of predicted and observed chlorophyll during the first half of the year. Later bloom levels are underestimated by the model, which may be due to (1) an overestimation of the flushing rate, (2) a violation of the assumption that no living phytoplankton are taken in, or (3) an overestimation of the background extinction, to which this energy-limited system is highly sensitive.  

9.4.1.11. Zoommeer. Since the Zoommeer (Figs. 9.17 and 9.18) does not exist yet, it is impossible to use observed lake-specific inputs to validate the model. This is no problem for some inputs such as temperature, solar intensity, or depth, but without an integrated version of CHARON and BLOOM II, we can only speculate about future nutrient concentrations. Because the Zoommeer will receive water similar to that of the Haringvliet, the nutrient concentrations of the latter have been used as the best available estimates. Also the value of the background extinction of the Haringvliet has been adopted, but because it can be varied easily, the sensitivity to a value of 2.5 instead of 1.2 per m has also been investigated (Fig. 9.18).

With these assumptions BLOOM II computes (extremely) high potential phytoplankton levels in all parts of the year no matter whether $K_b$ is 1.2 or 2.5 per m. Using Haringvliet inputs, all limiting factors are important at least some part of the year and Microcystis seems
Fig. 9.16--Stuwpand Lith, 1976: Predicted and observed chlorophyll
in mg/m³; nominal run
Fig. 9.17--Zoomeer, 1976: Predicted chlorophyll in mg/m³; inputs taken from Haringvliet
Fig. 9.18--Zoommeer, 1976: Predicted chlorophyll in mg/m³; all inputs taken from Haringvliet, but background extinction 2.5 m⁻¹
the most likely dominant species, but many other groups, including (dino)flagellates and diatoms dominate frequently in the simulations.

It must be stressed that these simulations are not intended to predict bloom levels in the future Zoommeer accurately, but rather to give an order of magnitude. Interpreted this way, the computations indicate that severe bloom problems similar to those in the Randmeren are likely in the Zoommeer, unless the nutrient concentrations are reduced significantly below the Haringvliet levels. With the integrated version of BLOOM II and CHARON, a more sophisticated prediction will be possible.

9.4.2. Overall Conclusions for Nominal Runs

9.4.2.1. Validation of BLOOM II. Considering that BLOOM II was calibrated to the drinking water reservoir Grote Rug and applied for PAWN without adjusting the values of the universal inputs, it produces very reasonable results. In most cases size, time, and limiting factors of objectionable blooms are well or reasonably well predicted. For the IJsselmeer, Veluwemeer (1975 and 1976), Wolderwijd (1975 and 1976), Haringvliet, and Stuwpand Lith, there was excellent agreement between predictions and observations. The results for the Gooimeer, Slotermeer, and Westeinder are fair, and in the last lake even excellent in the second half of 1976. Results for the Markermeer (in the second quarter) and IJmeer (all year) are rather poor. Usually the dominant phytoplankton group is well computed, but sometimes the wrong species is predicted, e.g., Microcystis instead of Oscillatoria. It would have been relatively simple to improve the predicted species compositions, but this was not attempted because no information on the PAWN lakes was to be used in recalibrating the model.

BLOOM II tends to predict higher than observed total chlorophyll concentrations, because it maximizes total biomass. Large overpredictions can occur if:

- The mortality rate constants are higher than the minimum estimate of the model.
- The amount of one or more of the limiting resources is overestimated, for example, because a large organic or inorganic particulate fraction is measured and erroneously assumed to be available to phytoplankton.
- The carbon to chlorophyll ratios exceed the (universal) values used in the model, which are on the lower end of the literature range.
- The steady state assumptions are violated, for example, when nutrient concentrations or the background extinction fluctuate significantly over short periods of time.

Frequently, however, there is reasonable agreement between predicted and computed biomass concentrations. Thus there is reason to be
confident about the model's capabilities to make realistic estimates of maximum potential blooms in future situations.

9.4.2.2. General Conclusions for the Existing Situation. The details of eutrophication in the Netherlands are highly time and site specific. Different biomass concentrations, limiting factors, and bloom compositions are expected and indeed computed by BLOOM II in different lakes and for different parts of the year. For each of the main constraints (nitrogen, phosphorus, and energy), there is at least one lake in which it is more important than any other factor. But for the cases considered here, nitrogen and energy are more frequently limiting than phosphorus. Also the computed yearly maxima (almost always during summer) are usually nitrogen or energy, rather than phosphorus, limited.

9.5. FIXATION OF ATMOSPHERIC NITROGEN

Some species of blue-greens can, under certain conditions, develop so-called heterocysts, which enable them to use atmospheric nitrogen N₂ rather than nitrate or ammonia. Among these are two of the model's species: Aphanizomenon and Anabaena. Although it was shown in Table 9.1 that many blooms are nitrogen limited in eutrophic Dutch lakes, nitrogen fixation seems a rare process. According to Zevenboom et al. [9.9, 9.10], studying phytoplankton in the Wolderwijd, and Horne [9.11], this must be due to the large energy requirement of this process.

To test whether (1) the model indeed computes small amounts of N fixers, and (2) these amounts remain low under simulated future conditions, the nominal set of species was extended with an N-fixing strain of Aphanizomenon. This species differs from the normal species in its requirement for N, which is zero, and in its maximum production rate, which is taken as 45 percent of the nominal value,⁶ based upon data by Stewart [9.12].

According to the model, N fixation is only possible in three lakes under nominal conditions: (1) Veluwemeer, (2) Wolderwijd, and (3) IJmeer. Typically N fixers contribute less than 50 percent to total biomass and only during a relatively short period of the year (the end of spring and first part of summer). The increase in total biomass is usually insignificant.

The reason the N-fixing strain dominates only infrequently is its high energy requirement. Thus it can only bloom in shallow lakes in periods with a (severe) nitrogen limitation.

It is expected that nitrogen will become a less important limitation in the future, since the most promising eutrophication control tactics reduce the available amounts of phosphorus or energy rather than nitrogen. Hence the probability of nitrogen fixation decreases. For example, Barica et al. [9.13] observed a significant shift to
non nitrogen-fixing species such as Microcystis and even flagellates, when they added nitrogen to a lake dominated by the N-fixing Aphanizomenon.

NOTES

1. Some additional runs using a version of the latest model have confirmed that, in spite of some changes in the results, the main conclusions remain unaltered.
2. The nutrient fraction, which BLOOM II allocates to living phytoplankton and detritus is called "planktonic"; the difference between this amount and what is totally available is called "rest"; this is the "slack" of the optimum LP solution.
3. BLOOM II did compute dominance of Oscillatoria over Microcystis in the year 1977, which is not included in this report.
4. No flushing and only a 10-percent decrease in $K_b$ are sufficient to prevent underprediction in the second part of the year.
5. This value seems reasonable considering that background extinction and depth tend to be inversely related (Table 8.3).
6. With an increase of $P_{max}$ to 70 percent, essentially the same conclusions still hold.

REFERENCES


PART III: THE DISSOLVED OXYGEN MODEL (OXYMOD)
Chapter 10
INTRODUCTION TO THE OXYGEN MODEL

An inventory of regional water quality problems in the Netherlands in February 1979 [10.1] has indicated that low oxygen concentrations occur in a number of Dutch natural waters, including some lakes, due to the high oxygen demands of processes such as mineralization of organic matter and respiration by phytoplankton. Dissolved oxygen concentrations of less than 50 percent of saturation in these lakes are mainly caused by huge phytoplankton blooms. First, large quantities of dead organic matter are produced when a bloom collapses. Bacteria mineralize this organic matter, consuming oxygen in the process. Second, phytoplankton show a diurnal pattern of production and respiration. During the day there is a net oxygen production, but during the night there is only oxygen consumption. The higher the standing stock of the phytoplankton, the stronger will be the undersaturation of oxygen during the night. Large fluctuations in the dissolved oxygen destabilize the ecosystem.

In Dutch lakes, the external load of organic matter from wastewater discharges is usually rather small compared to the internal load caused by phytoplankton blooms. The relative importance of the contributions of both loads to the biochemical oxygen demand in these lakes is in agreement with this. The reverse will generally be the case in streams and channels, as is described in Vol. V of the PAWN reports.

Thermal stratification is another phenomenon with a large potential impact on the oxygen budget of a lake. Stratification does not occur frequently in Dutch lakes, because they are very shallow (less than 5 m). Nevertheless, stratification just above the sediment surface, which probably occurred in 1976, may result in an "explosive" release of nutrients from the bottom into the overlying water, as was discussed in Part I of this volume.

As mentioned in Sec. 1.1, low oxygen concentrations may cause the death of organisms (fish kills), bad odors, and mobilization of nutrients from the bottom sediments, and may finally lead to a severe deterioration of the ecosystem. Under these conditions the value of a lake for recreational purposes (e.g., swimming and fishing) will be seriously reduced.

Because of these rather dramatic impacts on water quality, there is a need to model the dissolved oxygen budget in Dutch lakes. With a model, management strategies and tactics to improve the water quality with respect to the oxygen budget in lakes can be evaluated. Part III, the present part of this volume, deals with the description of the dissolved oxygen model OXYMOD, its calibration, and its application to a number of selected Dutch lakes.
SYMBOLS USED FOR OXYMOD

\( a \quad \text{Stoichiometric ratio of oxygen consumed}
\]
and nitrogen oxidized \( \text{mg } O_2/\text{mg org-N} \)

\( X \quad \text{Phytoplankton biomass concentration} \quad \text{g/m}^3 \text{ or } \text{g/m}^2 \)

\( b \quad \text{Stoichiometric ratio of nitrate reduced}
\]
and oxygen consumed \( \text{mg/mg } O_2 \)

\( \text{BOD}_{5} \quad \text{Biochemical oxygen demand after five days}
\]
and at a temperature of 20 deg C \( \text{mg } O_2/l \)

\( c \quad \text{Stoichiometric ratio of oxygen consumed}
\]
and carbon oxidized \( \text{mg } O_2/\text{mg org-C} \)

\( O \quad \text{Dissolved oxygen concentration} \quad \text{mg } O_2/l \)

\( C_{Cl} \quad \text{Chloride concentration} \quad \text{mg Cl/l} \)

\( C_{R} \quad \text{Reareration constant} \quad \text{s}^2/(\text{m.d}) \)

\( E \quad \text{Temperature coefficient in mineralization rates} \quad \text{s}^2/(\text{m.d}) \)

\( F_{a} \quad \text{Phytoplankton contribution to oxygen budget} \quad \text{mg } O_2/(1.d) \)

\( H \quad \text{Average depth of the lake} \quad \text{m} \)

\( k \quad \text{Integration constant with indexes 1 to 6} \quad \text{1/d} \)

\( K_{2} \quad \text{Reareration transfer coefficient} \quad \text{1/d} \)

\( K_{L} \quad \text{Liquid film transfer coefficient} \quad \text{1/d} \)

\( K_{b} \quad \text{Bioturbation rate constant} \quad \text{1/d} \)

\( K \quad \text{Mineralization rate constant} \quad \text{1/d} \)

\( L_{C} \quad \text{Organic carbon from waste as oxygen equivalents} \quad \text{mg } O_2/l \)

\( L_{N} \quad \text{Organic nitrogen from waste} \quad \text{mg N/l} \)

\( N \quad \text{Nitrate concentration} \quad \text{mg N/l} \)

\( P \quad \text{Atmospheric pressure} \quad \text{mmHg} \)

\( P_{w} \quad \text{Vapour pressure of water in atmosphere} \quad \text{mmHg} \)

\( P_{a} \quad \text{Average gross production of phytoplankton} \quad \text{g } C/(\text{m}^2\text{h}) \)

\( Q_{i} \quad \text{Input flow} \quad \text{m}^3/\text{d} \)

\( R_{a} \quad \text{Average respiration by phytoplankton} \quad \text{g } O_2/(\text{m}^2\text{h}) \)

\( R_{b} \quad \text{Background bottom oxygen respiration} \quad \text{g } O_2/(\text{m}^2\text{d}) \)

\( R_{c} \quad \text{Weight ratio of carbon and chlorophyll in}
\]
phytoplankton \( \text{mg } C/\text{mg Chl} \)

\( t \quad \text{Time} \quad \text{d or h} \)

\( T \quad \text{Temperature} \quad ^{\circ}\text{C} \)

\( v \quad \text{Flow rate} \quad \text{m/s} \)

\( V \quad \text{Volume of the lake} \quad \text{m}^3 \)

\( W_{10} \quad \text{Wind speed at 10-m height} \quad \text{m/s} \)
Superscripts

w - water
b - bottom
' - diurnal variation
20 - at 20 deg C

Subscripts

l - living
d - dead
b - bioturbated
i - input flow (or week number in \( A_i \))
o - conc. at \( t=0 \)
s - saturation
l - organic carbon (waste or in phytoplankton)
n - organic nitrogen (waste)

REFERENCE

Chapter 11
THE OXYGEN MODEL (OXYMOD)

11.1. OBJECTIVES AND MODEL OUTPUT

The development of any model should start off with a problem analysis and a determination of objectives for the use of the model, because these will mainly determine its structure and output. Most of the problem analysis has been outlined in the introduction and can be summarized as follows:

- Phytoplankton blooms, and to a smaller extent wastewater discharges, give rise to low dissolved oxygen concentrations in lakes and a progressive deterioration of natural water ecosystems. The effects will be the more severe the larger the size of the bloom or the faster it collapses. Mineralization of organic matter from phytoplankton blooms, as well as diurnal fluctuations in oxygen production, is important to the oxygen budget.

Phenomena related to thermal stratification have been omitted from this problem definition, because not enough time was available to incorporate them in the oxygen model. But stratification, as we have stated before, is not very important in the shallow Dutch freshwater lakes.

The objectives for our dissolved oxygen model are:

- The model should give insight into the interrelation of the relevant processes affecting the dissolved oxygen budget.
- The model should predict the dissolved oxygen concentration in lakes in relation to phytoplankton dynamics (eutrophication) and biologically degradable matter.
- The model should offer information about the success of strategies and tactics for improving water quality with respect to the oxygen budget.

The output of the model must be in accordance with these objectives. In addition, the output must enable calibration to measurements. The combination of these two requirements leads to the following desired outputs:

- The weekly (daily) average oxygen concentration.
The daily maximum and minimum oxygen concentrations due to diurnal variations in net oxygen production by phytoplankton.

- The minimum oxygen concentration due to a collapsing phytoplankton bloom and the duration of eventual anaerobic (oxygen-free) conditions.
- The nitrate concentration during anaerobic conditions.

Some additional remarks must be made. First of all, the nitrate concentration has been added to the list of outputs, because it is used as an electron acceptor instead of oxygen when mineralization occurs under anaerobic conditions. Its inclusion makes it possible to continue the calculations after the onset of anaerobic conditions and to compute the duration of anaerobiosis. Second, the list of outputs is smaller than it was originally in Ref. 11.1, because the output with respect to a potential stratification and the probability that it will last long enough to cause an anaerobic hypolimnion have been omitted.

11.2. THE WORST CASE CRITERION

There will always be some uncertainty in model predictions, because a model is an imperfect representation of reality. Certain less important processes are not considered. Important processes, which are included in the model, must often be described in an imperfect and simplified way, for lack of knowledge or ideal input data.

It may happen, therefore, that the model predicts one of the system variables to have reached a critical value. We shall consider a situation critical if a particular system variable exceeds a threshold, below or above which a discontinuity or relatively rapid change occurs in at least one of the other system variables. As an example, consider a sudden, massive fish kill, which can take place when the dissolved oxygen concentration drops below a certain threshold for some time period.

Obviously a water quality manager is particularly interested in preventing such critical situations. Thus it is very important to prevent underestimations of water quality problems in model simulations, because even a highly infrequent exceedance of a threshold of one of the forcing system variables, such as the dissolved oxygen concentration, will cause severe and long-lasting deterioration of the ecosystem. Therefore we ought to choose our assumptions for constructing the dissolved oxygen model in such a way that it will always simulate the worst case instead of the actual case. This implies that we exchange some accuracy for the certainty that problems will not be significantly worse than simulations show.
11.3. STRUCTURE AND MAIN ASSUMPTIONS OF OXYMOD

The model is based upon a dynamic oxygen model for rivers developed by Simonsen and Harremoes [11.2]. To apply it to lakes, the advection term in their model has been omitted. The following processes are incorporated in the model:

- Input and output flow.
- Reaeration (exchange of oxygen between water and air).
- Mineralization of organic carbon and organic nitrogen from wastewater discharges to carbon dioxide and nitrate.
- Production and respiration of phytoplankton.
- Mineralization of dead phytoplankton in the water and at the bottom to carbon dioxide.
- Background bottom respiration.

The main structure of the model is represented by Fig. 11.1, which indicates the interrelation of the relevant processes. The lake is assumed to be completely mixed (mixing is mainly driven by the wind). This is a reasonable assumption, because in Dutch lakes the retention time of a few months to a year is much longer than the characteristic mixing time, which is a few weeks or less. However, it must be stressed that local oxygen concentrations may be lower (or higher) than the average values due to inhomogeneities in phytoplankton production or locally high external BOD loads. The model does not account for this.

As another important assumption, the volume of a lake is kept constant, implying that the input and output flows are the same. Over the years this is correct, but there may be quite a difference between water levels in summer and winter. However, errors caused by changes in volume tend to be small, since water levels usually change slowly, and since the errors are damped by other processes such as reaeration. Rain and evaporation are not included in the model, because their contributions to the oxygen budget can usually be neglected.

Phytoplankton are assumed to be the only oxygen-producing and-consuming organisms of importance. All other organisms (e.g., zooplankton and fish) are neglected. This is reasonable, as biomass concentrations and rates of production of other organisms in Dutch lakes are usually negligible compared with those of phytoplankton. No phytoplankton are assumed to be present in the input flow, as phytoplankton biomass concentrations are usually much smaller in streams or channels than in lakes. Besides, phytoplankton in running waters are usually of different species than in lakes, and we assume they die when they enter a lake. Phytoplankton biomass in the input flow is considered as a part of the measured BOD load.

To simulate the worst possible conditions for the oxygen budget in case of a suddenly collapsing bloom, we assume that all phytoplankton die instantly and do not settle to the sediment. This will indeed
Fig. 11.1—Block diagram of the dissolved oxygen model
lead to the worst case, because the mineralization rate and hence the oxygen demand in the water is higher than on the bottom. In reality, phytoplankton will die more gradually and sedimentation will take place to a certain extent.

11.4. THE MATHEMATICAL MODEL

11.4.1. The Dissolved Oxygen Equation

We shall start with the complete differential equation, which is essentially the dissolved oxygen mass balance. The individual terms, each representing one or more processes, are elucidated afterward.

\[
\frac{dO}{dt} = \text{SUM} \left[ \frac{Q_i (O_{i=0})}{V} + K_2 (O_{s=0}) - K_1 L - \sum_{i} a K_n L_n + F_a - R_b / H \right]
\]  

(11.1)

The individual terms represent, respectively: the input/output flow, aeration, mineralization of organic carbon and nitrogen from wastewater discharges, the resultant phytoplankton production, respiration, and mineralization, and the background bottom respiration. Symbols are explained below and in the list of symbols. We will now discuss the individual terms.

11.4.1.1. Input and Output Flow. As previously explained, water input and output flows are equal, which makes it possible to use either of them in the oxygen equation. Usually the input rather than the output flows are known, and hence we shall use the former.

11.4.1.2. Reaeration. Two of the parameters in the term describing the reaeration rate must be elucidated, the reaeration transfer coefficient \( K_2 \) and the saturation concentration of oxygen \( O_s \). The exchange of oxygen between water and air is rate limited by the liquid phase, and therefore dependent only on the liquid film transfer coefficient.

\[
K_2 = K_L / H
\]  

(11.2)

According to Refs. 11.3 and 11.4, the following equations describe the temperature and wind speed dependency of \( K_1 \) in a (semi-)stagnant lake:

...
\[
K_L = C_r \left(1.016 \right)^{T-20} 10^{2} \quad \text{for} \quad w_{10} > 2 \text{ m/s} \\
K_L = 0.1 \quad \text{for} \quad w_{10} < 2 \text{ m/s}
\]

(11.3)

The reaeration constant \( C_r \) has a value that varies between 0.03 and 0.06, mainly depending on the morphological characteristics of the lake. The saturation concentration, depending on temperature, partial vapor pressure, and chloride concentration (actually the ionic strength), can, according to Ref. 11.5, be computed as:

\[
Q_s = (0.68 \times 10^{-4} T) \times (P - P_w) \times (1 - 9 \times 10^{-6} c_{cl}) / (T+35)
\]

(11.4)

The partial vapor pressure of water is a function of temperature and can be described as:

\[
P_w = 4.6 + 0.032T^2
\]

(11.5)

Normally, however, the atmospheric pressure \( P \) equals 760 mmHg, which is much higher than \( P_w \), and the chloride concentration in freshwater lakes is very low. Thus we may simplify Eq. (11.4) to:

\[
Q_s = (0.68 \times 10^{-4} T) P / (T+35)
\]

(11.6)

11.4.1.3. Mineralization of Organic Carbon from Wastewater Discharges. The concentration of organic carbon \( L_c \) is expressed as oxygen equivalents. The rate constant \( K_L \) is a function of temperature and decomposer (bacteria) concentration. The latter in its turn is a function of the substrate concentration. To incorporate this functional dependence in the model, a decomposer compartment should be introduced. This was omitted because of the impossibility of calibrating the decomposer concentration, and its unknown relation to phytoplankton dynamics, but probably we would not have achieved a higher degree of accuracy. Thus only a temperature dependence is built into the model, using the following simple equation from Refs. 11.6 and 11.7:

\[
K_L = K_{L}^{20} E^{(T-20)}
\]

(11.7)

\( K_L^{20} \) is the rate constant at a temperature of 20 deg C, while \( E \) is the temperature coefficient.
11.4.1.4. Mineralization of Organic Nitrogen from Wastewater Discharges. The concentration of organic nitrogen $L_n$ is expressed as Kjeldahl N (mg/l). A discussion analogous to the one given above could be repeated here, concerning the temperature and decomposer concentration dependency of the rate constant $K_n$.

We must make an additional remark about the mineralization of organic nitrogen. Actually this process involves two completely independent steps: breakdown of organic nitrogen to ammonium and oxidation of ammonium to nitrate. The latter process is called nitrification and is accomplished by a different group of bacteria than the former. However, these processes are treated as one process by the model, because the first step, which proceeds much more slowly than the second, is rate controlling. This is supported by the observation that ammonium concentrations in lakes are usually low compared to nitrate concentrations. The stoichiometric constant $a$ in Eq. (11.1), which is the ratio of mg oxygen consumed and mg nitrogen oxidized, is equal to 4.57.

11.4.1.5. Phytoplankton Production, Respiration, Sedimentation, and Mineralization. The overall contribution of phytoplankton to the oxygen budget may be described as follows:

$$F_a = c \times dX/dt$$ (11.8)

which only states that the contribution of phytoplankton equals the change in total living and dead phytoplankton biomass times a stoichiometric constant $c$, the number of grams of oxygen produced (or consumed) per gram of organic carbon produced (or oxidized). The value for $c$ is 2.67. The change in total phytoplankton biomass is described as:

$$dX/dt = dX_w^{a}/dt + dX_d^{a}/dt + 1/H \times dX_d^{b}/dt +$$

$$1/H \times dX_b/dt + \text{SUM} Q_i (X_w^{a} + X_d^{a})/V$$ (11.9)

The first two terms are considered constant in an integration step. The concentration of living algae $X_w^{a}$, and the change in concentration in an integration step (one week), can be calculated by the phytoplankton model BLOOM II (Part II of this volume) or from measured chlorophyll data. The concentration of dead phytoplankton, $X_d^{a}$, is calculated from the concentration of living phytoplankton using a steady state equation, which was used in BLOOM II when OXYMOD
was applied for PAWN. In a later version of BLOOM II, this equation has been replaced (Secs. 7.3.2 and 9.1). The next term in Eq. (11.9) was calculated as:

\[ X_d^w = 0.0295 \times 1.066^T X_e^w / (0.006T + 0.15) \]  \hspace{1cm} (11.10)

As in BLOOM II, it is implicitly assumed that dead and living phytoplankton in the water are at steady state. Hence BLOOM's output can be fed directly into OXYMOD, which was our intent. As an additional implication, mineralization of dead phytoplankton in the water does not have to be described explicitly, which may be considered an advantage. The third term in Eq. (11.9) represents the change in the concentration of dead phytoplankton on the bottom, which have settled from the detritus pool of the water. The concentration of dead phytoplankton on the bottom can be calculated according to the next differential equation:

\[ \frac{dX_b^d}{dt} = H^wS^wX_d^w - K_{b1}^bX_d^b - K_b^bX_d^b + \]

\[ (dX_b^w/dt + dX_e^w/dt) \]  \hspace{1cm} (11.11)

The sedimentation rate constant S is equal to 0.15 d^{-1}, and K_{b1}^b is the mineralization rate constant of dead phytoplankton on the bottom, which is a function of temperature, analogous to Eq. (11.7). The third term in Eq. (11.11) indicates the disappearance rate of dead phytoplankton into the deeper layers of the sediment, mainly due to bioturbation. It is assumed that this part is mineralized very slowly. The oxygen demand of this slow process is considered as part of the background bottom oxygen demand.

If a bloom declines during an integration interval, the decrement in living and dead phytoplankton pools is assumed to migrate to the sediment. The last two terms (between parentheses) in Eq. (11.11) are then negative. Otherwise they are omitted.

The pool of bioturbated phytoplankton is not explicitly mineralized in the model. Its increase during an integration interval must therefore be added to the net contribution of phytoplankton to the oxygen budget. This is represented by the fourth term in Eq. (11.9), and thus:

\[ \frac{dX_b}{dt} = K_bX_d^b \]  \hspace{1cm} (11.12)
Presumably the bioturbation rate constant is a function of temperature. However, $K_b$ has a constant value in the model, because the temperature function is unknown. The last term in Eq. (11.0) gives the outflow of living and dead phytoplankton, which is small compared to the other terms.

11.4.1.6. The Background Bottom Respiration. According to Refs. 11.7 and 11.8, the oxygen consumption by the bottom can be computed as:

$$ R_b = R_{b29}E(T-20) $$

(11.13)

Literature values for $R_{b29}$ range from 0.5 to 2.0 g $O_2$/m$^2$d; a representative value is 1.0 g $O_2$/m$^2$d. This value may be too high in our model, because we treat the oxygen demand of dead phytoplankton on the bottom separately, although actually it is a part of the measured literature values for $R_{b29}$.

11.4.2. The Diurnal Variation of Oxygen

During the day, phytoplankton produce more oxygen by photosynthesis than they consume by respiration. During the night only respiration of organic matter proceeds, and there is net consumption of oxygen. As a result of this diurnal rhythm, a minimum oxygen concentration is observed in the early morning just before dawn, and a maximum occurs late in the afternoon. The diurnal variation from the daily average oxygen concentration $O'$ can be calculated from:

$$ \frac{dO'}{dt} = -(\text{SUM } Q_i/V + K_2)O' + F_a' $$

(11.14)

Equation (11.14) has been derived by subtracting the daily average value of the oxygen concentration from Eq. (11.1). This is permitted only if the daily average value is constant during the integration interval (e.g., a week), which is indeed the case in a first order approximation. The diurnal variation in the average net oxygen production by phytoplankton is equal to the difference between the hourly net production and its daily average value as a function of time:

$$ F_a' = F_a' - R_a' $$

(11.15)
The hourly net production is calculated by BLOOM II from a
distribution of the daily net production over the day.

11.4.3. Organic Carbon and Nitrogen from Wastewater Discharges

Organic carbon and nitrogen can be calculated from two simple
differential equations:

\[
\frac{dL_c}{dt} = \text{SUM } \frac{Q_i(L_{ci} - L_c)}{V} - K_2L_c \tag{11.16}
\]

\[
\frac{dL_n}{dt} = \text{SUM } \frac{Q_i(L_{ni} - L_n)}{V} - K_1L_n \tag{11.17}
\]

The balances have only input and output flows and one decay term. \(L_c\)
and \(L_n\) cannot be used for calibration of the model, because they
have not been measured separately. Only total BOD\textsubscript{5}\textsuperscript{20} and
Kjeldahl N, which includes phytoplankton, have been measured.
Nevertheless \(L_c\) and \(L_n\) can serve as additional information about
the oxygen-consuming processes and their relative importance.

11.4.4. Decay of a Suddenly Collapsing Bloom

Equation (11.1) can also be used to compute the dissolved oxygen
concentration if a bloom suddenly collapses. The phytoplankton
contribution, however, has to be calculated according to:

\[
F_a = -c(K_1^w X_d^w + K_1^b X_d^b / H) \tag{11.18}
\]

In agreement with the worst case criterion, we assume that all
phytoplankton die instantly and become part of the detritus pool in
the water. The mineralization rate constant \(K_1^w\) has the same value
as \(K_1^b\). We assume that no more sedimentation of dead phytoplankton
takes place following the collapse, so the equation for dead
phytoplankton on the bottom is changed to:

\[
\frac{dX_d^b}{dt} = - K_1^b X_d^b - K_b^b X_d^b \tag{11.19}
\]
Following a collapse the oxygen concentration starts to decrease rapidly. If it drops below 0.5 ppm, nitrification stops due to a lack of oxygen and $K_n$ is set to zero. At this point we consider the water to be anaerobic, and use nitrogen as an electron acceptor in the mineralization process in addition to oxygen. The reduction of nitrate to elemental nitrogen is called denitrification. The change of the nitrate concentration is represented by:

$$\frac{dN}{dt} = \text{SUM}_i \left( \frac{Q_i (N_i - N)}{V} + b \times dO/dt \right)$$

The second term represents the denitrification rate and is proportional to that part of the oxygen demand which cannot be supplied by oxygen itself. This term is negative while nitrate is consumed, and can be calculated from Eq. (11.1) with a $K_n$ of zero, multiplied by a stoichiometric constant $b$ (the ratio of mg nitrate consumed and mg oxygen consumed) of 2.92.

According to Ref. 11.6, the mineralization rate by denitrification is probably lower than that of aerobic mineralization. However, because this rate is highly uncertain, and in agreement with the worst case criterion, we have used the same rate constants for both processes.

When the second term of Eq. (11.20) becomes positive again, the concentration of oxygen begins to rise. Once it rises above 0.5 ppm, nitrate is no longer consumed and the model returns to the oxygen equation. The elapsed time between downward and upward passing of the 0.5-ppm threshold for the oxygen concentration is the duration of anaerobic conditions.

11.4.5. Solution of the Differential Equations

Each of the previously described differential equations can be solved analytically. The solutions for Eqs. (11.11), (11.12), (11.14), (11.16), (11.17), and (11.19) are straightforward and consist of at least one exponential term and a constant. The solution of the oxygen Eq. (11.1) is derived by substituting the solutions for $L_c$ and $L_h$ for the mineralization terms and the solutions of Eqs. (11.11) and (11.12) for the phytoplankton contribution $F_a$. The ultimate solution of the oxygen equation has the following form:
\[
0 = \left( o_0 + \frac{k_1}{k_2} - \frac{k_3}{(k_4-k_2)} - \frac{k_5}{(k_6-k_2)} \right) * \exp(kt) + \\
\frac{k_3}{(k_4-k_2)} * \exp(kt) + \\
\frac{k_5}{(k_6-k_2)} * \exp(kt) - \frac{k_1}{k_2} 
\]

(11.21)

The same procedure with a similar result should be applied to solve the nitrate Eq. (11.20).

11.4.6. Model Inputs

OXYMOD, which has now been developed, requires four different types of inputs: (1) constants, (2) input flow variables, (3) in-lake variables, and (4) atmospheric variables. These are gathered in Table 11.1.

We must make some additional remarks about the input of organic carbon from wastewater discharges, \( L_c \). Biodegradable organic carbon in natural water is very frequently measured as biochemical oxygen demand during 5 days at 20 deg C. Obviously this \( \text{BOD}_{5}^{25} \) is not equal to the ultimate oxygen demand, because after 5 days some undegraded organic carbon is left. However, this problem can be solved, as the ultimate oxygen demand can be computed from the \( \text{BOD}_{5}^{25} \) value using:

\[
L_c = \frac{\text{BOD}_{5}^{25}}{1 - \exp(-5*K_1)} 
\]

(11.22)

\( K_1 \) is the decay rate under laboratory conditions at 20 deg C, and equals 0.23 d\(^{-1}\).

Another complication may arise if phytoplankton interfere with the \( \text{BOD}_{5}^{25} \) method, because for some time they continue to respire in the dark until eventually they die. This problem will not be discussed any further, but we must stress that the \( \text{BOD}_{5}^{25} \) figures may be inaccurate for our purposes.

11.4.7. The Computer Code OXYMOD

The computer code OXYMOD consists of a main routine for input, output, and some basic calculations and three subroutines: OXMEAN, MAXMIN, and DECAY. Subroutine OXMEAN calculates the daily average oxygen
Table 11.1
SUMMARY OF INPUTS FOR THE DISSOLVED OXYGEN MODEL

<table>
<thead>
<tr>
<th>Type of Input</th>
<th>Description of Parameters</th>
<th>Symbol</th>
<th>Dimension</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constants</td>
<td>Mineralization rate</td>
<td>$K_n$</td>
<td>$d^{-1}$</td>
</tr>
<tr>
<td></td>
<td>constants at 20 deg C</td>
<td>$K_w, K_b$</td>
<td>$d^{-1}$</td>
</tr>
<tr>
<td></td>
<td>Bottom oxygen respiration</td>
<td>$R_b$</td>
<td>$d^{-1}$</td>
</tr>
<tr>
<td></td>
<td>Temperature coefficients</td>
<td>$E$</td>
<td>$-$</td>
</tr>
<tr>
<td></td>
<td>for these 5 rates</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Stoichiometric constants</td>
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<td>mg/mg</td>
</tr>
<tr>
<td></td>
<td>Reaeration constant</td>
<td>$C_r$</td>
<td>$s^2/(md)$</td>
</tr>
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<td>Sedimentation rate constant</td>
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<td>$m^3/d$</td>
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</tr>
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<td>$BOD_5$,</td>
<td>$L_{ci}$</td>
<td>$mg/l$</td>
</tr>
<tr>
<td></td>
<td>Kjeldahl N, and nitrate</td>
<td>$L_{ni}$</td>
<td>$mg/l$</td>
</tr>
<tr>
<td>In-lake</td>
<td>Volume of lake</td>
<td>$V$</td>
<td>$m^3$</td>
</tr>
<tr>
<td>variables</td>
<td>Lake surface area</td>
<td>$O$</td>
<td>$m^2$</td>
</tr>
<tr>
<td></td>
<td>Temperature</td>
<td>$T$</td>
<td>$^\circ C$</td>
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<tr>
<td></td>
<td>Concentrations of nitrate</td>
<td>$N$</td>
<td>$mg/l$</td>
</tr>
<tr>
<td></td>
<td>and living phytoplankton</td>
<td>$X_{1}^w$</td>
<td>$mg C/l$</td>
</tr>
<tr>
<td></td>
<td>Deviation hourly net prod.</td>
<td>$F_a$</td>
<td>$mg C/l$</td>
</tr>
<tr>
<td>Atmospheric</td>
<td>Wind speed</td>
<td>$W_{10}$</td>
<td>$m/s$</td>
</tr>
<tr>
<td>variables</td>
<td>Air pressure</td>
<td>$P$</td>
<td>mmHg</td>
</tr>
</tbody>
</table>

The time interval is a week, but can easily be changed if necessary. Thus OXMEAN computes the daily average oxygen concentration in the middle of each week. In addition it calculates the concentrations of organic carbon (as oxygen equivalents) and nitrogen from wastewater discharges and the concentrations of dead and bioturbated phytoplankton at the bottom (as g C/m²).

Subroutine MAXMIN computes the daily maximum and minimum oxygen concentrations. It calculates the hourly deviation from the daily average oxygen concentration and selects the highest and lowest values. These are added to or subtracted from the average daily oxygen concentration to obtain the diurnal maximum and minimum.
Subroutine DECAY simulates the oxygen concentration after a sudden collapse of a bloom in any previously selected week. The time interval is a day. In addition it calculates the nitrate concentration when the oxygen concentration drops below 0.5 ppm and the concentrations of organic carbon and nitrogen, and bioturbated dead phytoplankton on the bottom.

REFERENCES


Chapter 12

CALIBRATION OF OXYMOD

The oxygen model was calibrated for Ring 3 of the storage reservoir Grote Rug in 1976. The input dataset consisted of weekly in-lake data. As mentioned in Sec. 1.2.2, Ring 3 receives untreated water from the Wantij, a branch of the Rijn. The input flow data from the Wantij were collected from Ref. 12.1, the quarterly water quality reports of the Rijkswaterstaat. The atmospheric inputs were obtained from a nearby weather station of the KNMI (Royal Dutch Meteorological Institute). The weekly average wind speed was computed as the quadratic mean from the daily values.

12.1. CALIBRATION SETUP

The biomass levels for the calibration were computed from the measured chlorophyll levels, which were then converted to phytoplankton carbon concentrations. For the first 29 weeks of the year the conversion ratio \( R_c \) of carbon to chlorophyll in phytoplankton was set at 30, because flagellates and diatoms were the dominant groups of species. For the remainder of the year \( R_c \) was set at 40 because the blue-green species *Microcystis* was dominant. These conversions are similar to those used by BLOOM II and shown in Table 8.2.

In contrast to the lake simulations (Chap. 13), a volume correction for evaporation and precipitation was necessary for the Ring 3 calibration, because volume varied substantially (by more than 30 percent).

In the first series of calibrations, dissolved oxygen concentrations fluctuated rather strongly with unrealistic peak levels and dips. These fluctuations are due to the simple way in which the phytoplankton contribution \( F_a \) is described in the model. Therefore we decided to smooth the phytoplankton biomass input data according to the following equation:

\[
A_i = \frac{(X_{i-1} + 2X_i + X_{i+1})}{4}
\]

(12.1)

Using this equation, we reduce the effect of strong phytoplankton biomass fluctuations to an acceptable level, and the results of the model improve significantly.

To calibrate the model, it was necessary to adopt two different temperature functions for each temperature coefficient \( E \) in Eq. (11.7). Relatively low values of \( E \) (hence high mineralization rates) were used
at temperatures above 12 deg C, and relatively high values of \( E \) (hence low mineralization rates) were used below 12 deg C. There is sufficient evidence in Refs. 12.2 and 12.3 that indeed mineralization rates decline rapidly at temperatures below 12 deg C. The values adopted during the calibration and used for the model simulations are shown in Table 12.1. Obviously some of the rates have now become discontinuous, which may be theoretically wrong, but which we think causes no unacceptable errors in practice.

With the above-mentioned adjustments of the model, its sensitivity was evaluated for variations in:

- The mineralization rate constants at 20 deg C and the temperature coefficients within reasonable ranges.
- The respiration constant between 0.03 and 0.06.
- The background bottom respiration rate constant at 20 deg C and the temperature coefficient.
- The bioturbation rate constant.

12.2. DISCUSSION OF THE CALIBRATION RESULTS

The final calibration results are shown in Fig. 12.1, in which measured and calculated oxygen concentrations are compared. Although the model does not quite fit the extremes, it still follows the trend in the measurements quite satisfactorily. The parameter values which were varied during calibration, and whose final values resulted in the oxygen curve of Fig. 12.1, are gathered in Table 12.1.

Some remarks must be made about the measured oxygen concentrations. The measured values are not entirely representative for two reasons. First, measurements were not made at the right time of day, which is between 12 a.m. and 1 p.m. (Sec. 11.4.2), when the oxygen concentration approximates its daily average value. Second, the oxygen concentration is not constant in the vertical direction, high values being observed near the surface and low values near the bottom. Although in Grote Rug measurements were made at three different depths, the computed average is not equal to the actual average, because sometimes the difference between the maximum and minimum over the vertical is very large.\(^1\)

Another part of the difference must be due to the way phytoplankton biomasses are calculated from chlorophyll data. The carbon to chlorophyll ratio \( R_c \) is assumed to be constant, which probably is not the case. It was mentioned earlier (Sec. 8.2.1) that variations in this ratio due to changes in biotic and abiotic conditions may be quite significant.

The remaining differences can largely be attributed to the model itself. A major weakness is the way the phytoplankton contribution \( F_a \) to the oxygen budget has been represented. Considering it a constant within a one-week time interval is an oversimplification, but
Fig. 12.1--O₂ concentration in Ring 3 of De Grote Rik in 1976:
Final calibration results
is inevitable due to lack of information on the dynamic behavior of phytoplankton.

Finally, it must be stressed that Ring 3 was extremely hard to simulate correctly in 1976. The nutrient budgets and phytoplankton dynamics were unstable in comparison with other years and other rings. Nevertheless, the 1976 data for Ring 3 represented the best available dataset by the time OXYMOD was calibrated. For all these reasons, we consider differences of 1 ppm between measured and calculated dissolved oxygen concentrations to be insignificant.

Table 12.1
PARAMETER VALUES ESTABLISHED BY CALIBRATION

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Process</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mineralization of organics</td>
<td>$K_{1}^{28}$</td>
<td>0.2 d$^{-1}$</td>
</tr>
<tr>
<td>waste discharges</td>
<td>$E$</td>
<td>1.047 -</td>
</tr>
<tr>
<td>Mineralization (oxid.) of</td>
<td>$K_{n}^{28}$</td>
<td>0.1 d$^{-1}$</td>
</tr>
<tr>
<td>organic nitrogen</td>
<td>$E$</td>
<td>1.08 -</td>
</tr>
<tr>
<td>from waste discharges</td>
<td>$E$</td>
<td>1.12 -</td>
</tr>
<tr>
<td>Mineralization of</td>
<td>$K_{1}^{28b}$</td>
<td>0.1 d$^{-1}$</td>
</tr>
<tr>
<td>dead phytoplankton</td>
<td>$E$</td>
<td>1.047 -</td>
</tr>
<tr>
<td>on the bottom</td>
<td>$E$</td>
<td>1.10 -</td>
</tr>
<tr>
<td>Background</td>
<td>$R_{b}^{28}$</td>
<td>1.0 g/(m$^2$d)</td>
</tr>
<tr>
<td>bottom respiration</td>
<td>$E$</td>
<td>1.07 -</td>
</tr>
<tr>
<td></td>
<td>$E$</td>
<td>1.20 -</td>
</tr>
<tr>
<td>Bioturbation</td>
<td>$K_{b}$</td>
<td>0.0143 d$^{-1}$</td>
</tr>
<tr>
<td>Reaeration</td>
<td>$C_{r}$</td>
<td>0.045 s$^2$/m$^2$</td>
</tr>
</tbody>
</table>

NOTE

1. Appropriately weighting the three measurements in calculating the average would probably give better results if only we had a good idea of what the weights should be. The weights depend upon the forms one expects the profile of oxygen concentration with depth to take, and our information is too sparse to make this determination.
REFERENCES

Chapter 13

DISSOLVED OXYGEN SIMULATIONS WITH OXYMOD

The dissolved oxygen model, developed and calibrated as described in the preceding chapters, has been applied to the eleven PAWN lakes (Sec. 1.4) under current conditions. All oxygen computations are for the extremely dry year of 1976. As in the case of BLOOM II, it must be stressed that OXYMOD was not recalibrated for the PAWN lakes. With the exception of the reaeration constant $C_r$, only nominal parameter values have been used. In order to cope with the unexpectedly high reaeration rate in the natural lakes, due to a larger surface and hence higher waves than in Ring 3 of Grote Rug, the reaeration constant $C_r$ was set to the upper limit found in literature of 0.06. Thus, we may view as validation cases the simulations for the PAWN lakes under the conditions of 1976.

Both measured and computed (with BLOOM II) phytoplankton biomass concentrations have been used in the simulations. These are referred to as cases A and B, respectively, in this chapter's figures. Measured chlorophyll was converted to carbon using values for $R_c$ determined by linear regressions of chlorophyll to several particulate nutrient fractions (Sec. 8.2.1). These are shown in Table 13.2. Notice that the output of BLOOM II used for the nominal cases of OXYMOD is not exactly the same as the BLOOM II output discussed in Part II of this volume. As mentioned in Chap. 9, BLOOM II was considerably improved after its application for PAWN, and all nominal results have been recomputed. However, the conclusions of Chap. 15 are not affected by these differences in phytoplankton computations.

13.1. DATA ACQUISITION

It was very difficult to obtain a complete, yet accurate, set of input data, and a major part of them had to be estimated. Obviously, this has implications for the accuracy of the simulation results.

The sources of the inputs for each lake are gathered in Table 13.1 with the attached code explanation. A few remarks must be made about the nature of these data. Data on in-lake variables, and on oxygen and $BOD_5$ concentrations in inflowing water, were usually available as two or four weekly time series and were linearly interpolated to obtain weekly data. The quantities of inflowing water, however, were not known exactly, except those of the Maas River. They were computed either from a model (IJsselmeer and Haringvliet) or from average values for the period 1969-1974. A distinction has been made between winter and summer half-years. The input flows were assumed to be constant within these half-year periods. The weekly average wind speeds have been calculated as the mean square of the daily values of one or two
### Table 13.1

**SOURCES OF INPUT DATA FOR LAKE SIMULATIONS**

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>IJsselmeer</td>
<td>a) Ketelmeer(IJssel)</td>
<td>1</td>
<td>6</td>
<td>6</td>
<td>6</td>
<td>De Kooi</td>
</tr>
<tr>
<td></td>
<td>b) Polders</td>
<td>2</td>
<td>2</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Markermeer</td>
<td>Polders</td>
<td>2</td>
<td>2</td>
<td>6</td>
<td>6</td>
<td>De Kooi</td>
</tr>
<tr>
<td>IJmeer</td>
<td>a) Gooimeer</td>
<td>2</td>
<td>6</td>
<td>6</td>
<td>6</td>
<td>De Kooi</td>
</tr>
<tr>
<td></td>
<td>b) Noordzeekanaal</td>
<td>2</td>
<td>6</td>
<td></td>
<td></td>
<td>De Bilt</td>
</tr>
<tr>
<td></td>
<td>b) Vecht</td>
<td>2</td>
<td>7</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Veluwemeer</td>
<td>a) 2 Waterpur. plants</td>
<td>2</td>
<td>2</td>
<td>6</td>
<td>6</td>
<td>Eelde/</td>
</tr>
<tr>
<td></td>
<td>b) Polders</td>
<td>2</td>
<td>2</td>
<td></td>
<td></td>
<td>De Bilt</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>Polders</td>
<td>2</td>
<td>2</td>
<td>6</td>
<td>6</td>
<td>Eelde/</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>De Bilt</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>a) Eem</td>
<td>2</td>
<td>7(Eembr)</td>
<td>6</td>
<td>6</td>
<td>Eelde/</td>
</tr>
<tr>
<td></td>
<td>b) Wolderwijd</td>
<td>2</td>
<td>6(V106)</td>
<td></td>
<td></td>
<td>De Bilt</td>
</tr>
<tr>
<td>Westeinder-</td>
<td>Ringvaart</td>
<td>3</td>
<td>3(41)</td>
<td>3</td>
<td>8</td>
<td>LH-Rott.</td>
</tr>
<tr>
<td>plassen</td>
<td>Haarlemmermeer</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Slotermeer</td>
<td>Ee/Luts/Polders</td>
<td>9</td>
<td>9</td>
<td>4</td>
<td>4</td>
<td>Lelyst./</td>
</tr>
<tr>
<td></td>
<td>HaarlemmermeerLeeuw</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Haringvliet</td>
<td>Hollandsch Diep</td>
<td>1</td>
<td>6(H9)</td>
<td>6</td>
<td>6</td>
<td>Vliss./</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>LH-Rott.</td>
</tr>
<tr>
<td>Stuwpand Lith</td>
<td>Maas Grave</td>
<td>5</td>
<td>6</td>
<td>6(Lith)</td>
<td>6</td>
<td>Volkjel</td>
</tr>
<tr>
<td>Zoommeer (Kommeer)</td>
<td>Hollandsch Diep</td>
<td>1</td>
<td>6(H9,H17)</td>
<td>-</td>
<td>-</td>
<td>Vliss./</td>
</tr>
</tbody>
</table>

**NOTES:** The numbers refer to works listed below. Codes between parentheses refer to measurement locations. Wind speed data are obtained from the KNMI.

---

**Number code for Table 13.1**

5. Anon., *Jaarverslag 1976, Rijncommissie Waterleidingbedrijven (RIWA)*.
weather stations. Additional remarks about the lake-specific inputs are tabulated in Table 13.2.

Concerning the measured dissolved oxygen concentrations in the lakes, comments analogous to those given in Sec. 12.2 hold true. Oxygen concentrations have been measured every two or four weeks, so it is quite possible that some extreme peaks or dips were overlooked.

13.2. DISCUSSION OF NOMINAL SIMULATION RESULTS

All results of the nominal simulation runs are graphically presented in Figs. 13.1 through 13.21. For each lake except Zoommeer, we made two simulations: case A, using measured phytoplankton concentrations, and case B, using predictions from BLOOM II. For each simulation we report the diurnal variation in the oxygen concentrations and the impacts of a collapsing bloom, because these simulations are started in those weeks having the highest chlorophyll levels. Because BLOOM II tends to overpredict phytoplankton biomasses (Sec. 9.4.2), the case B simulations usually show greater impacts of a collapse than do the case A simulations.

These simulations can be viewed both as validations of OXYMOD and as assessments of the present (as of 1976) conditions of the PAWN lakes. In the first view, attention should be directed to how well the calculated oxygen concentration matches the observed concentration, which is what the figures show. In the second view, it is more important to note how wide the diurnal fluctuation is, and whether and for how long anaerobic conditions may persist after a collapse of a bloom. We defer our discussion of these latter aspects of the OXYMOD simulations to Chap. 16.

13.2.1. IJsselmeer (Figs. 13.1; 13.2)

Measured and calculated oxygen concentrations match well in both cases A and B. Undoubtedly, this is due to the very stable behavior of the phytoplankton biomass concentration. One measurement (week 23) lies rather far above the calculated curve, which may be explained as follows. Chlorophyll was measured in weeks 19 and 23, and hence values for the intermediate weeks were obtained by linear interpolation. The actual phytoplankton biomass concentration may have been totally different, and we may have skipped a peak in biomass. On the other hand, a rapid shift in the carbon to chlorophyll ratio could be a partial explanation for it, too.

The maximum diurnal variation is 2 ppm in summertime. Usually oxygen problems are absent. Serious oxygen problems may occur, however, if a bloom collapses between weeks 15 and 40. The model predicts anaerobic conditions with a duration of at most 12 days.
Table 13.2
ADDITIONAL COMMENTS TO THE INPUT DATA
FOR LAKE SIMULATIONS

<table>
<thead>
<tr>
<th>Lake</th>
<th>Comments on the Input Data</th>
<th>( R_c )</th>
</tr>
</thead>
<tbody>
<tr>
<td>IJsselmeer</td>
<td>- ( Q_2 ) from polders estimated from the water balance 1969-1974 as 8 percent of ( Q_1 ) from the Ketelmeer. ( L_n_2 ) and ( N_2 ) are estimated as respectively 5/12 and 7/12 of total N. ( L_{c2} ) was estimated as the yearly average of ( L_{c1} ) ( O_2 ) was estimated to be the saturation conc.</td>
<td>30</td>
</tr>
<tr>
<td>Markermeer</td>
<td>- Estimation procedure for the loads coming from the polders was the same as for the IJsselmeer. Water exchange between the Markermeer, IJsselmeer, and IJmeer is unknown; we ignored it.</td>
<td>30</td>
</tr>
<tr>
<td>IJmeer</td>
<td>- ( L_n ) in the Vecht is estimated.</td>
<td>40</td>
</tr>
<tr>
<td>Veluwemeer</td>
<td>- Estimation procedure for the loads coming from the polders was the same as for the IJsselmeer. ( L_c ) and ( L_n ) in the discharges from the purification plants estimated with the assumption that 1 inhabitant equivalent is 54 g BOD/d and 13 Kj-N/d. ( O_2 ) in these loads was estimated to be 5 ppm.</td>
<td>40</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>- Estimation procedure for the loads coming from the polders was the same as for the IJsselmeer.</td>
<td>65</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>- ( L_n ) in the Eem input flow is estimated.</td>
<td>40</td>
</tr>
<tr>
<td>Westeinder-</td>
<td>- The input flow was assumed to come entirely from the Ringvaart Haarlemmermeer, equal to 2 m³/s.</td>
<td>40</td>
</tr>
<tr>
<td>plassen</td>
<td>Slotermeer</td>
<td>- Inputs considered as one input flow, calculated from quarterly balances. ( L_n_1 ) and ( N_1 ) are estimated as respectively 5/12 and 7/12 of total N, ( L_{c1} ) as ( 2* L_{n1} ). ( O_1 ) was estimated to be the saturation conc. ( T ) was assumed to have the values of the Veluwemeer.</td>
</tr>
<tr>
<td>Haringvliet</td>
<td>- No chlorophyll data available for weeks 1 to 20. They were estimated by linear interpolation.</td>
<td>30</td>
</tr>
<tr>
<td>Stuwand</td>
<td>- ( L_c ) in the input flow from the Maas river was corrected for phytoplankton biomass.</td>
<td>30</td>
</tr>
<tr>
<td>Lith</td>
<td>Zoommeer</td>
<td>- The loads of the Eendracht are unknown. They are assumed to be the same as those of the Hollandsch Diep. The loads from the Hollandsch Diep were those of 1976. Its ( Q_1 ) was equal to 21 m³/s, a minimum value for desalinization of the Zoommeer. The initial values of in-lake variables have been taken from the Haringvliet in 1976.</td>
</tr>
</tbody>
</table>
Fig. 13.1--Simulation oxygen model, IJsselmeer, 1976 (case A)
Fig. 13.2—Simulation oxygen model, IJsselmeer, 1976 (case B)
13.2.2. Markermeer (Figs. 13.3; 13.4)

The agreement between measured and calculated oxygen concentrations is satisfactory in both cases A and B. The underprediction in week 26 may be explained by the same causes that were discussed for the IJsselmeer.

The maximum diurnal variation is 2 ppm. Chlorophyll levels in case A are only about a quarter of those in the IJsselmeer, which explains why no serious oxygen problems are predicted at a collapsing bloom. The simulated oxygen concentrations in case A never drop more than slightly below 50-percent saturation (min. 3.6 ppm). Although the predicted biomass levels are substantially higher than observed, the duration of anaerobic conditions is less than one day for case B.

13.2.3. Veluwemeer (Figs. 13.5, 13.6)

Although the measured and calculated oxygen curves match to an acceptable degree, underprediction is quite substantial between weeks 24 and 26. It is remarkable that the measured chlorophyll level decreases, while the oxygen concentration goes up. This strongly suggests a shift in the carbon to chlorophyll ratio. Overprediction occurs in the last quarter of the year. It is clear that OXYMOD tends to predict less well when phytoplankton levels fluctuate strongly, as in the Veluwemeer.

The maximum diurnal variation is 6 ppm in summertime. This results in a minimum daily oxygen concentration slightly under 50-percent saturation (4.5 ppm), which is undesirably low.

Potential oxygen problems are predicted if a bloom collapses between weeks 15 and 45. Although chlorophyll levels are much higher, problems are less dramatic than in the IJsselmeer, because reaeration better compensates the oxygen consumption by mineralization in the shallow Veluwemeer. Still, the predicted anaerobic conditions have a maximum duration of 7 days in case A and of 15 days in case B. Even nitrate depletion occurs.

13.2.4. Wolderwijd (Figs. 13.7; 13.8)

The measured and calculated oxygen curves match well during the first half of the year in case A. Again an underprediction is observed in week 26. Case B is much less successful because the predicted phytoplankton biomasses of the former version of BLOOM II were not well synchronized with the observations. Most remarks for the Veluwemeer also apply to the Wolderwijd.

The maximum diurnal variation is 8 ppm in summertime, which implies daily minima below 50-percent saturation. The oxygen concentration drops to 4.2 ppm in week 34 in the early morning.
Fig. 13.3--Simulation oxygen model, Markermeer, 1976 (case A)
Fig. 13.4—Simulation oxygen model, Markermeer, 1976 (case B)
Fig. 13.5--Simulation oxygen model, Veluwemeer, 1976 (case A)
Fig. 13.6--Simulation oxygen model, Veluwemeer, 1976 (case B)
Fig. 13.8--Simulation oxygen model, Wolderwijd, 1976 (case B)
Oxygen problems are predicted if a bloom collapses between weeks 15 and 45. They are much more severe than in the Veluwe meer due to an even higher biomass (in terms of carbon, not in terms of chlorophyll). Anaerobic conditions have a maximum duration of two weeks, while nitrogen is depleted for more than a week.

13.2.5. Gooimeer (Figs. 13.9; 13.10)

Oxygen concentrations are strongly overpredicted and underpredicted between weeks 10 and 18 in case A. This must be due to a very high, suddenly appearing chlorophyll peak in week 11, with which the model cannot cope, as we stated before. Due to the more stable character of the predicted phytoplankton biomass, the results for case B are much better.

The maximum diurnal variation is 4.6 ppm, causing low oxygen concentrations between weeks 25 and 29 with a lower bound of 3.5 ppm. If a bloom collapses between weeks 15 and 25, OXYMOD predicts anaerobic conditions with a duration of maximally 7 days in case A and of 14 days in case B.

13.2.6. IJmeer (Figs. 13.11; 13.12)

Dissolved oxygen concentrations are overpredicted between weeks 9 and 13 in both cases A and B, which must be due to the very high and suddenly appearing biomass peak in week 10 (Fig. 9.11). In case A there is some underprediction in summertime, which may be caused by a shift in the carbon to chlorophyll ratio.

The maximum diurnal variation is 2 ppm. The same remarks as for the Markermeer can be made for a collapsing bloom. Here potential problems are even smaller.

13.2.7. Slotermeer (Figs. 13.13; 13.14)

The measured oxygen concentrations are a little scattered around the calculated curve A, which may be because all measured values are from only one location, which is not very representative for the lake. Case B shows some overprediction, due to overpredicted phytoplankton biomass levels. The maximum diurnal variation is 4.4 ppm.

In case A, low oxygen concentrations of 4 to 5 ppm during a week are predicted if a bloom were to collapse. In case B, anaerobic conditions are predicted lasting 6 days or less.

13.2.8. Westeinderplassen (Figs. 13.15; 13.16)

Although some overprediction occurs in the last quarter of the year, simulations A and B are both satisfactory. Diurnal variation is 3 ppm or less.
BASIS=GEOMETEN

Fig. 13.9--Simulation oxygen model, Gooimeer, 1976 (case A)
Fig. 13.11 -- Simulation oxygen model, IJmeer, 1976 (case A)
Fig. 13.12--Simulation oxygen model, IJmeer, 1976 (case B)
Fig. 13.13—Simulation oxygen model, Slotermeer, 1976 (case A)
Fig. 13.14--Simulation oxygen model, Slottermeer, 1976 (case B)
No anaerobic conditions occur after a collapse, because obviously chlorophyll levels are too low. The lowest value achieved after a collapse was 3 ppm.

13.2.9. Haringvliet (Figs. 13.17; 13.18)

In case A, oxygen concentrations are overpredicted between weeks 10 and 12, probably because no chlorophyll data were measured for the first 20 weeks. Simulation B is not successful, because the calculated chlorophyll levels were too high\(^2\). The maximum diurnal variation is 2.0 ppm. A collapse of the blooms in late spring or summer could result in 7 days of anaerobic conditions.

13.2.10. Stuw pand Lith (Figs. 13.19; 13.20)

Unlike the previously discussed lakes, the Stuw pand is not actually stagnant, but a river segment with relatively low flow rates of 0.05 to 1.0 m/s. Therefore an additional equation must be used for the reaeration transfer coefficient as in Ref. 13.1:

\[
K_2 = 5.326 \left( \frac{v}{0.67} \right)^{-0.85} H^{0.3} \quad 0.03 < v < 1.5 \text{ m/s} \quad (13.1)
\]

This function is governed by the flow rate instead of the wind speed. Eq. (13.1) is used only if its value exceeds the value of the original wind speed function (Eq. (11.3)).

As for the model results, one should recall that the model was constructed for a stagnant lake and not for a river segment in which the water has a considerable flow rate. There is hardly any longitudinal mixing, and the average retention time is only one day. Reaeration, sedimentation, and phytoplankton production will proceed differently in a river compared to a lake.

Hence measured and calculated oxygen concentrations agree poorly. One may only conclude from Figs. 13.19 and 13.20 that the model cannot cope with flowing rivers, as might have been expected.

13.2.11. Zoommeer (Fig. 13.21)

The Zoommeer is a lake which will be constructed in the near future. Only the shallow part near Bergen op Zoom, which will be called the "Kommeer," has been simulated. Obviously all input data had to be estimated, and the calculation results are rather speculative.

Figure 13.21 shows the oxygen curve predicted from the (very high) biomass levels computed by BLOOM II. Oversaturation occurs all year. The diurnal variation may be as high as 10 ppm.
Fig. 13.17--Simulation oxygen model, Haringvliet, 1976 (case A)
Fig. 13.18--Simulation oxygen model, Haringvliet, 1976 (case B)
Anaerobic conditions could arise and persist for one week if a bloom were to collapse at any time of the year. Hence the impacts are less dramatic than might be expected for a lake with phytoplankton biomasses as high as those predicted for the Kommeer. The reason is that reaeration is relatively quick in this shallow lake.

13.3. SUMMARY OF NOMINAL SIMULATION RESULTS

The relevant results of the simulations are summarized in Table 13.3. Obviously, oxygen problems are strongly related to the degree of eutrophication. The higher the biomass levels and the lower the stability of the ecosystem, the worse oxygen problems may be, especially in case a bloom collapses. High bloom levels also result in large diurnal fluctuations in dissolved oxygen, which under extreme conditions can have a minimum below 50-percent saturation.

However, the severity of oxygen problems is not strictly proportional to the phytoplankton biomass peaks, because the average depth of a lake is also important. In relatively deep lakes, reaeration affects the oxygen concentration more slowly, and larger fluctuations are possible. Compare, for example, the IJsselmeer (depth 4.5 m) to the Veluwemeer (depth 1.2 m). Differences in depths are particularly important to the impacts of a collapsing bloom, but less important to the diurnal variation of the oxygen concentration.

<table>
<thead>
<tr>
<th>Lake</th>
<th>Daily Average Oxygen Conc.</th>
<th>AbsOLUTE Oxygen Concentration</th>
<th>Maximal Diurnal Variation</th>
<th>Maximal Anaerobiosis</th>
<th>Maximal Phytoplankton Biomass</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Min.</td>
<td>Max.</td>
<td>ppm O₂</td>
<td>d</td>
<td>(mg C/l)</td>
</tr>
<tr>
<td>IJsselmeer</td>
<td>8.8</td>
<td>14.6</td>
<td>7.5</td>
<td>15.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Markermeer</td>
<td>7.8</td>
<td>14.2</td>
<td>7.0</td>
<td>14.3</td>
<td>2.0</td>
</tr>
<tr>
<td>IJmeer</td>
<td>7.4</td>
<td>16.5</td>
<td>6.5</td>
<td>16.8</td>
<td>2.0</td>
</tr>
<tr>
<td>Veluwemeer</td>
<td>7.2</td>
<td>15.5</td>
<td>4.3</td>
<td>16.1</td>
<td>6.0</td>
</tr>
<tr>
<td>Wolderwijd</td>
<td>8.0</td>
<td>15.7</td>
<td>4.2</td>
<td>15.9</td>
<td>8.0</td>
</tr>
<tr>
<td>Gooimeer</td>
<td>5.7</td>
<td>20.0</td>
<td>3.5</td>
<td>20.1</td>
<td>4.6</td>
</tr>
<tr>
<td>Westeinder</td>
<td>8.3</td>
<td>14.2</td>
<td>7.1</td>
<td>14.3</td>
<td>3.0</td>
</tr>
<tr>
<td>Slotemeer</td>
<td>8.4</td>
<td>14.6</td>
<td>6.5</td>
<td>15.1</td>
<td>4.4</td>
</tr>
<tr>
<td>Haringvliet</td>
<td>6.7</td>
<td>14.6</td>
<td>5.8</td>
<td>15.0</td>
<td>2.0</td>
</tr>
<tr>
<td>St. Lith</td>
<td>6.1</td>
<td>15.0</td>
<td>5.3</td>
<td>15.5</td>
<td>2.0</td>
</tr>
<tr>
<td>Zoommeer</td>
<td>10.8</td>
<td>22.0</td>
<td>6.9</td>
<td>24.3</td>
<td>10.0</td>
</tr>
</tbody>
</table>

NOTES: The absolute minimum and maximum oxygen concentrations are the yearly minima and maxima computed for any diurnal cycle. The maximum duration of anaerobic conditions occurs when a bloom suddenly collapses.
It is also obvious from the results that the maximum concentrations shown in Table 13.3 occur in winter, when the dominance of phytoplankton production over mineralization results in oversaturation. In summer usually the opposite is observed, resulting in undersaturation and hence minimum concentrations.

Besides these trends, the results summarized in Table 13.3 also enable us to draw a few conclusions about the oxygen budget in the simulated Dutch freshwater lakes, which are discussed in Chapter 15.

NOTES

1. This asynchronization is much less noticeable in the current version of BLOOM II. For OXYMOD, however, the older BLOOM II results have been used.
2. The Haringvliet is another example where the current version of BLOOM II produces considerably improved results (see above, Fig. 9.15).

REFERENCE

Chapter 14

RECOMMENDATIONS FOR FUTURE RESEARCH

14.1. IMPROVEMENT OF THE DISSOLVED OXYGEN MODEL

The dissolved oxygen model can be improved in several respects. First, the way in which the phytoplankton contribution is described in the model could be improved if a dynamic rather than an equilibrium phytoplankton model were available to compute living and dead biomass concentrations and their sedimentation rates as functions of time.

Second living biomass concentrations are usually measured photometrically in terms of chlorophyll. To calibrate a (dynamic) phytoplankton model, its output must be compared to observed chlorophyll concentrations, which implies that the species-specific carbon to chlorophyll ratio $R_c$ should be modeled.

Little is known about the mineralization of sewage and dead phytoplankton. A number of aspects may have to be incorporated or improved in the oxygen model, of which the most important seem to be (1) the distinction of several organic fractions, including a refractory fraction, with different mineralization rates, (2) autolysis of dead phytoplankton, and (3) the relation of rates to temperature and chemical compositions.

In the model the bottom respiration depends only on temperature, while in reality it is also a function of the transport rate. The relation between the dispersion coefficient and bottom respiration should be investigated.

Although thermal stratification is not frequently observed in Dutch freshwater lakes, it is worthwhile incorporating it into OXYMOD. Its implications for water quality can be quite substantial. The explosive release of nutrients from the bottom after it becomes reduced may stimulate phytoplankton growth and hence affect the oxygen budget.

Another possible refinement of the model is introducing a two-step mechanism for the oxidation of organic nitrogen: the first for breakdown of organic nitrogen to ammonium, the second for the oxidation of ammonium to nitrate.

As it stands now, OXYMOD cannot cope with local high pollution or phytoplankton production levels. For these the model should consider different compartments within a lake. The same is true for vertical inhomogeneities arising from thermal stratification, buoyancy of phytoplankton, or very rapid, vertically varying processes. To incorporate these, the model should be extended with dispersive and advective transport phenomena.
14.2. IMPROVEMENT OF MEASUREMENTS OF CALIBRATION DATA

To evaluate an oxygen model, one needs a representative set of measured oxygen concentrations. As discussed in Sec. 12.2, such a dataset is not available at the moment. It would be useful to measure oxygen at the right time of day, that is, between 12 a.m. and 1 p.m., and at several depths, at least near the surface, at half the depth and near the bottom. Thus might a representative picture be obtained of the vertical oxygen profile and the average value. This is especially important in case of stratification. Continuous monitoring of the dissolved oxygen concentration during phytoplankton blooms would be useful to track the diurnal variation.

Finally, measuring the BOD$_5$ the way it is presently done should be a matter of discussion. The outcome of a BOD$_5$ measurement is the complex result of a number of processes, among which are respiration and biodegradation of phytoplankton, while for the modeling of the oxygen budget only the latter is desired.

14.3. IMPROVEMENTS IN PREDICTING IMPACTS OF TACTICS

Tactics which reduce the concentrations of phytoplankton biomass or organic matter from wastewater discharges can be evaluated by OXYMOD in combination with a phytoplankton model. For PAWN this was done in combination with the steady state model BLOOM II. However, the combination of these two models is imperfect, because BLOOM II calculates potential maxima, which could differ strongly from the actual concentrations. Even if the phytoplankton predictions are correct for a time-step, the oxygen computations may be wrong if the derivative of the computed phytoplankton time series is incorrect. This is a problem since BLOOM II tends to predict biomasses more smoothly than actually observed. The course of a time series, especially when it fluctuates strongly, is decisive for eventual oxygen problems. It would thus be an advantage to have a (dynamic) model to compute phytoplankton biomass as a function of time. With such a model we could better evaluate the impacts of eutrophication control tactics.
Chapter 15

CONCLUSIONS FROM OXYMOD VALIDATION RUNS

The following conclusions may be drawn from the calibration and simulation runs. Concerning the nominal results:

1. The oxygen budget in Dutch freshwater lakes is governed mainly by phytoplankton dynamics. The biochemical oxygen demand of mineralization of organic matter from wastewater discharges plays a minor role.
2. The 50-to-150-percent provisional standard for oxygen saturation is violated in Veluwemeer, Wolderwijd, and Gooimeer.
3. The 80-to-120-percent target standard for oxygen saturation is violated in all lakes.
4. Severe oxygen depletion problems following the sudden collapse of a bloom occur in all simulated lakes except Markermeer, IJmeer, and Westeinderplassen.
5. A safe chlorophyll standard in Dutch freshwater lakes to prevent oxygen depletion after a collapse would be a maximum (not average) concentration between 50 and 100 mg/m³.

Concerning the dissolved oxygen model OXYMOD:

1. The model gives generally satisfactory results.
2. Because measured oxygen concentrations are not completely representative, model deviations of 1 ppm O₂ or less have little meaning.
3. OXYMOD in its present form cannot deal well with lakes which have high and strongly fluctuating phytoplankton biomass levels.
4. OXYMOD could be improved using a more dynamic description for the phytoplankton contribution.
5. OXYMOD in its present form gives no information about the consequences of local high BOD loads or vertical inhomogeneities. It should be extended with transport phenomena to make compartmentizing possible.

Concerning measurements of calibration data:

1. The dissolved oxygen concentration should be measured at the right time of day, between 12 a.m. and 1 p.m., and at several representative depths.
2. The meaning of the present BOD$_5$ measurement should be a matter of discussion, at least with respect to phytoplankton respiration.
PART IV: RESULTS AND CONCLUSIONS
Chapter 16
EVALUATION OF EUTROPHICATION CONTROL TACTICS

16.1. LIMITS TO THE CURRENT APPROACH

We have applied BLOOM II to investigate the impacts of several eutrophication control tactics on phytoplankton blooms in the lakes considered for PAWN. In addition, we have applied OXYMOD to evaluate the impacts on the oxygen budgets and the risks of anaerobic conditions. A complete analysis of eutrophication control tactics was impossible, however, because of certain data and model limitations: as pointed out in Sec. 1.3, we were unable to complete a model that could compute future nutrient concentrations in the lakes from future nutrient discharges into the national water network.

Partly related to the foregoing, it was impossible to investigate all interactions of tactics. Usually the purpose of a tactic is to reduce the available amount of an essential resource such as a nutrient or energy for a significant period. But many tactics might produce side effects, which could either increase or decrease their total effectiveness. For example, it follows from Sec. 3.1.2 that any bloom-reducing tactic could minimize the probability of an explosive bottom flux of phosphorus, since less nitrate is incorporated by phytoplankton, hence more remains dissolved.

Because these and similar interactions could not be quantified, tactics were investigated as if they would not produce any side effects. For example, when we considered the tactic of dredging to increase the depth, we assumed that all nutrient concentrations and the background extinction would remain the same as in the nominal case.

We investigated each tactic by parametrically increasing its effect on the relevant environmental constraint(s). The most extreme simulated conditions are intended to be an upper bound on the impacts a tactic might have if it were actually implemented. However, for some tactics (e.g., homogeneous mixing or phosphate removal) it is much easier to establish a realistic upper bound than for others (e.g., incrementing the background extinction).

The effectiveness of tactics is indicated by their impacts on the yearly maximum phytoplankton concentrations in mg dry weight per m³. This index was selected rather than chlorophyll because the ratio of dry weight to chlorophyll is bound to change if (1) the compositions of the blooms and (2) the average light climate change significantly. Variations in chlorophyll due to changes in this ratio do not reflect an actual improvement or degradation of the eutrophic state of a lake, which changes in dry weight concentrations too. To indicate how the dry weight predictions are related to the chlorophyll standards, a boundary for 50 and 100 mg chlorophyll per m³ is shown in all figures, based upon the presently existing variation in the dry weight to chlorophyll ratios between lakes and species.
We decided to consider maximum rather than average phytoplankton concentrations because (1) the potential impacts of a bloom are to a large extent determined by the maximum of a certain period (e.g., the larger a collapsing bloom, the more serious the oxygen depletion problem), and (2) because generally summer averages change similarly to yearly maxima. If the impacts of a tactic vary seasonally, this will be indicated in the text.

When comparing cases with a tactic to cases without (i.e., the nominal cases), one must always consider how well the nominal calculation for a lake agreed with the observations. The probability that the predicted impacts are correct declines the more BLOOM II nominally overpredicts. Suppose, for example, that the predicted year maximum in the nominal case is two times higher than observed and the predicted biomass reduction from a tactic is a factor of 1.5. In reality the tactic could be completely ineffective, or it might indeed reduce biomass by a factor of 1.5, depending on why the model overpredicts, whether or not it computes limitations correctly, which factor is affected by the tactic, etc.

It must also be pointed out that the impacts of tactics have only been assessed for 1976, which was an extremely dry and sunny year. Besides solar radiation, nutrient levels could be different in other years, for example because water in the network tends to be more diluted. An examination of 1975 and 1977 data, however, has indicated that the main conclusions would also hold for these years.

Obviously there are several uncertainties in the previously discussed assumptions. Also notice that the impacts of all but one tactic (homogeneous mixing) were analyzed using an earlier version of BLOOM II. Thus, conclusions should not be based on small differences between simulations. But in many lakes there is a considerable difference in the effectiveness of tactics, and many of the more general conclusions are very robust.

Notice that only the individual points in Figs. 16.1 through 16.4 correspond to actual simulations by BLOOM II, but for clearness the points were connected by drawn lines.

16.2. EUTROPHICATION CONTROL STANDARDS

For the "Nieuw Indicatief Meerjaren Plan" (new IMP), RIZA has proposed several standards related to eutrophication, for example for chlorophyll, total phosphorus, and dissolved oxygen. The "provisional" (marginally acceptable) standard for chlorophyll is 100 mg/m³, while
Fig. 16.2--Effect of increasing the background extinction coefficient on yearly maximum biomass predictions.
Fig. 16.3--Effect of flushing on yearly maximum biomass predictions
Fig. 16.4 -- Effect of dredging on yearly maximum biomass predictions
the "target" (ultimately preferred) standard is 50 mg/m³. The provisional standard for phosphorus is 0.3 mg/l,¹ and the target value is 0.05 mg/l. For dissolved oxygen, the provisional standard requires the concentration to remain between 50 and 150 percent of saturation, while the target value requires it to stay between 80 and 120 percent of saturation. Although these standards are reasonable as an average in an average lake, it is important to add several comments:

- The possible variation by a factor of 2 or 3 in the dry weight to chlorophyll ratios results in an uncertainty of a factor of 2 or 3 in the dry weight at which the chlorophyll standard is violated (Sec. 8.2.1): yet the environmental impact of this variation may be substantial.
- Because the chlorophyll standards are summer averages, considerable peaks (two, three, or more times the standard) are still possible. Large variations in phytoplankton biomass, however, have great ecological impacts (e.g., on the oxygen budget), and increase the probability of anaerobic conditions when the bloom declines.
- The chlorophyll and related eutrophication standards are not always consistent. As we shall explain in more detail in Sec. 16.3.2, the provisional standard for total phosphorus is too high, because it does not take into account that (1) current yearly maxima are not usually phosphorus limited and (2) blooms could shift in composition.
- Regional variations in functions and characteristics of lakes, which are quite substantial, are not taken into account.

In the remainder of this chapter, we shall regard the provisional and target values for chlorophyll not as acceptable averages, but as acceptable peak values. The existing variations in dry weight to chlorophyll will be indicated.

16.3. TACTICS INVOLVING AVAILABLE NUTRIENT CONCENTRATIONS

Presently nutrients in general and nitrogen in particular are important constraints to phytoplankton blooms in Dutch lakes. Tactics to reduce available nitrogen therefore look promising, but unfortunately, as pointed out in Vol. III, none of them seem technically and economically feasible. Thus only tactics involving available silicon and phosphorus have been investigated.

16.3.1. Nutrient Balancing: Silicon Addition

Many of the adverse symptoms of phytoplankton blooms are directly related to the dominance of blue-green species. Therefore it seems an improvement if the bloom composition could be changed, even if total biomass would not decline. One possible mechanism is addition of silicon to the water to favor growth of diatoms over blue-greens.
To investigate the effectiveness of this tactic we made a run for each lake in which the silicon concentration was kept constant at 5 mg/l, the highest value presently observed in any lake. This resulted in an increase in total biomass and a shift in favor of diatoms only in Veluwemeer, Wolderwijd, and Slotermeer. The effect always occurred in the second quarter of the year. There was no effect during other parts of the year or in any other lake, because diatoms are usually limited by energy rather than silicon as a consequence of the high extinction. Also the increase in total biomass in already eutrophic lakes could present problems. Therefore we concluded that silicon addition is not generally effective and could produce side effects.

16.3.2. Phosphate Removal

There are two important reasons why phosphorus removal is often proposed as the best available tactic to reverse eutrophication: (1) blooms in many oligotrophic, unpolluted lakes are limited by phosphorus, and (2) a considerable reduction in the phosphate discharge into rivers is technically feasible.

But investigating the effectiveness of P removal is much more complicated than sometimes suggested. First, it is difficult to predict future P loadings from rivers, because it is uncertain whether all significant P discharges, in particular those into the Rijn, can be reduced both in the Netherlands and in foreign countries. This problem is important considering that many lakes receive more water (and P) from this river than from any other source. For many Dutch lakes, three quarters of the annual P load originates from the Rijn.

Second, it is uncertain how the chemical systems of the lakes are affected by a significant reduction in P loadings. What happens to the fluxes to and from the bottom? As shown in Part I of this volume, a large percentage of incoming P is annually removed to the bottom, hence prediction of future concentrations in the water strongly depends on the question whether or not this percentage remains the same. The effectiveness of P removal would be significantly reduced if less of the P load were removed to the bottom, or if the "normal," annual flux from the bottom did not change (or increased), or if explosive fluxes kept occurring. On the other hand, the effectiveness could also be greater than expected, since any decrease in phytoplankton biomass may result in a rise of the dissolved nitrate concentration, which decreases the probability of explosive bottom fluxes of P.

Third, as shown in Table 9.1, presently most blooms are limited by nitrogen or energy rather than phosphorus, which has two important consequences: (1) to make P limiting, an often considerable amount has to be removed without a positive effect on bloom levels, but (2) when P becomes limiting, blooms may shift in composition from species that require little energy and nitrogen, such as Microcystis, to
species that require more energy and nitrogen but less phosphate, such as Aphanizomenon. Until blooms completely consist of species with low P requirements, any reduction in external P concentration by a certain percentage "x" reduces the biomass by less than "x" percent, because the bloom will be more and more composed of P-efficient species.

These biological impacts could be investigated with BLOOM II, but as was earlier remarked, it was not possible to relate nutrient concentrations in the lakes to discharges into rivers. Therefore the potential impacts on the PAWN lakes of P reductions were investigated by imposing maximum available P concentrations in the lakes for each week of the year. These concentrations were parametrically reduced until they were equal to the target standard for total P (0.05 mg/l).

The results for the considered lakes are shown in Fig. 16.1. For each lake there is a line in the figure with a dot at its right-hand end. The dot is positioned horizontally at the maximum total phosphorus level observed in the lake in 1976, which is the sum of dissolved phosphorus and phosphorus incorporated in phytoplankton. The dot is positioned vertically at the maximum phytoplankton biomass computed by the model for that lake in 1976. It may be observed that the upper right-hand section is horizontal for each lake, which indicates that the computed maximum is not limited by P. As we have mentioned previously, the amount of total available phosphorus should be reduced below the amount contained in the computed maximum bloom, before any result on bloom sizes may be expected.

The length of the horizontal section of the lines in Fig. 16.1 indicates how much P has to be removed before it becomes limiting. For example, we may predict that the IJsselmeer or Wolderwijd are much more susceptible to phosphate reduction than the Stuwand Lith or the Westeinder. However, in the lakes with the largest blooms, enormous percentages of phosphorus must be removed before the chlorophyll standards are met (roughly between 40 and 90 percent). For example, with a 50-percent reduction in total phosphorus, the basic chlorophyll standard might just be met in only two lakes (IJsselmeer and Markermeer), where computed bloom levels currently exceed this standard. But this is only true if we adopt the maximum biomass to chlorophyll ratio that currently exists.

Figure 16.1 also shows that a value of 0.3 mg/l is much too high as a provisional standard for total phosphorus (again, see Note 1), because calculated bloom levels are hardly different from the maxima currently observed. The target value of 0.05 mg/l is right in the middle of the zone which indicates a chlorophyll concentration of 50 mg/l; hence it seems about right to meet the target standard for chlorophyll.

It is also important to note that the predicted blooms in most lakes are still dominated by blue-greens, unless the current phosphorus maxima are reduced by 50 percent or more. The moment that a shift to less objectionable species occurs is mainly determined by the light
climate, because blue-greens tend to outcompete all other species if energy is frequently limiting.

16.4. TACTICS INVOLVING AVAILABLE ENERGY

Energy is an important constraining factor to the yearly maxima in some of the relatively deep lakes and the winter bloom levels in some lakes and also is highly important to the species dominance. Tactics which reduce the availability of energy thus seem an attractive alternative to other tactics such as P removal, particularly since they are effective regardless of nutrient concentrations.

However, the number of technically, economically, and ecologically feasible tactics is small. Thus of all possible suggestions only few have actually been investigated.

16.4.1. Incrementing the Background Extinction

Presently the background extinctions range from 1.2 m\(^{-1}\) in the Haringvliet to 6.5 m\(^{-1}\) in the Slotermeer. Increasing the background extinctions seems attractive because a rise of only 1.0 per m could reduce the bloom size at which energy becomes limiting by approximately 50 to 200 mg chlorophyll\(^{2}\) per m\(^3\). Whether indeed bloom levels are reduced by this amount depends of course on all limiting factors, including nutrients.

The background extinction, which affects the total amount of available energy, might also affect the spectral distribution of light in the water if some wavelengths are more strongly attenuated than others. This could be important to phytoplankton bloom levels and compositions, but we had insufficient information for a quantitative analysis.

Possible ways to increment the background extinction are discussed in Vol. III, where the use of colored dyes is suggested. Whether this is acceptable will not be discussed here, but only the potential impacts on bloom levels.

The results (Fig. 16.2) show that yearly maxima are greatly reduced in the IJsselmeer, Markermeer, Haringvliet, and Stuwpand Lith. They are reduced to a smaller extent in the Gooimeer, Zoommeer, and Westeinder. Yearly maxima are hardly reduced at all in any other lake. Because energy is more frequently limiting in winter, bloom levels are more strongly reduced in that part of the year.

The large difference in effectiveness between lakes is explained by the (exponential) attenuation of light in the water, which is determined by the product of the total extinction and depth. Thus incrementing the background extinction is most successful in lakes which are either deep, such as the Haringvliet, or have a relatively high nominal background extinction, such as the Gooimeer.
In conclusion, this tactic could be effective in some lakes, although unfortunately not in the most eutrophic.

16.4.2. Complete Mixing

Two groups of species, blue-greens and dinoflagellates, are considerably favored over other groups since they can regulate their vertical position in the water to a great extent. Concentrating in the surface layers, they can absorb more light than other, homogeneously distributed species and "overshadow" these. But even if no other species are present, buoyancy control still significantly increases the level at which energy becomes limiting, because it enables absorption of light quanta which would otherwise be absorbed by homogeneously distributed background and dead phytoplankton (Eqs. (7.24) and (7.29)). Because there is little quantitative information on the net benefit of buoyancy control, an empirical relation was established for WABASIM [16.1] using data on energy-limited blooms in Grote Rug. This same relation has been used in all bloom computations for PAWN.

However, it is important to remember that this factor was determined by calibration of the model. Hence we cannot rule out the possibility that what is computed as the benefit from buoyancy control in the model is (partly) due to some other factor. For instance, we might also underestimate the maximum production rate, or use respiration or mortality rates that are too high.

Water can be mixed artificially at least on a local scale as in the Biesbosch drinking water reservoirs. Whether similar control measures are possible elsewhere depends on the local circumstances and will not be discussed. To compute how much buoyancy control contributes to the success of blue-green phytoplankton (dinoflagellates are of minor importance on a national scale) and what the impacts would be if all lakes could be mixed homogeneously, we simulated all cases except Haringvliet and the Stuwpand Lith assuming no buoyancy control.

Complete mixing has been simulated in the model by removing the empirical advantage of blue-greens and dinoflagellates, which means they are treated as any other species in the model. Because for this tactic there was a significant difference between the results of the current and the former version of BLOOM II in two or three lakes, the impacts of complete mixing have been recomputed. Thus we observe some small differences between the nominal total biomass values in Table 16.1 and those shown in the figures for the other tactics.

As may be seen in Table 16.1, complete mixing effectively reduces the maximum total biomass in lakes which have either a large background extinction, a large depth, or both. In the deep Markermeer the predicted maximum bloom level is strongly reduced compared to the nominal case. In the IJsselmeer, which although deep has a low background extinction, and in the Gooimeer, Slotermeer, and
Westeinder, which have low depths but high background extinctions, maximum total biomass is reduced by about one-third to one-half. In the nominal cases, maximum blooms are strongly dominated by blue-green phytoplankton (except in the Zoommeer). Complete mixing reduces blue-greens substantially in all lakes, although they remain dominant in some of them.

### Table 16.1

<table>
<thead>
<tr>
<th>Lake</th>
<th>Nominal Simulations</th>
<th>Complete Mixing</th>
<th></th>
</tr>
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<tbody>
<tr>
<td></td>
<td>Total Biomass</td>
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<td>Total Biomass</td>
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</tbody>
</table>

These simulations are in agreement with observations by Lorenzen and Mitchell [16.2], who observed that *Aphanizomenon* blooms in a 3-m-deep lake became confined to a smaller part of the year, but achieved about the same peak levels with and without artificial mixing. When validated against the 1975 data of the artificially mixed Biesbosch reservoir "De 130," BLOOM II computed total biomasses and species compositions reasonably in agreement with the observations. It may be concluded that complete mixing seems to be an interesting tactic, which could be quite effective in reducing total biomass and the probability of blue-green blooms, particularly in deep or turbid lakes. It would be interesting to see whether these conclusions can be confirmed experimentally.

#### 16.4.3. Flushing

In rivers, where considerably more nutrients are available than in stagnant waters, phytoplankton concentrations are usually much lower. Thus in lakes flushing seems an attractive possibility for decreasing eutrophication problems. However, there are both quantity and quality problems. Enough water to reduce a lake's retention time significantly will be available only in a limited number of locations. Moreover, unless water taken in meets rather strict quality standards, increased flushing rates also imply increased annual loadings of many pollutants, including nutrients.
As explained earlier, it was impossible to compute nutrient concentrations under revised loading conditions with the existing eutrophication models. Therefore we had to assume that: (1) the intake water contains no living or dead phytoplankton, but only dissolved nutrients; (2) dissolved nutrients and those in living and dead phytoplankton are removed in proportion to their concentrations in the lake; and (3) flushing does not affect the total concentration of any nutrient in the water of a lake.

The first two assumptions seem rather trivial: river water, which would usually be used for flushing, contains mainly dissolved nutrients. Also there is no obvious reason why all three nutrient fractions considered in a lake would not be removed in proportion to their concentrations. The third assumption could be questioned. However, the flushing water will generally have higher nutrient concentrations than the lake, so assuming the same weekly total nutrient concentrations could well be optimistic.

The impacts of decreasing retention times on phytoplankton blooms vary significantly between lakes (Fig. 16.3). Flushing is effective in lakes with a relatively large depth or background extinction such as the IJsselmeer and Gooimeer, and yearly maxima can be reduced by more than 50 percent with retention times of 10 days. Winter bloom levels can even be virtually eliminated. In shallow lakes, however, yearly maxima increase at first and only start decreasing below the nominal value if the retention time becomes extremely short (about 5 days). Winter bloom levels decrease, but usually remain objectionably high.

These results can be explained by the two contradictory impacts of flushing, which:

- Increase the loss of living phytoplankton, similar to an increased mortality rate, which can be compensated only if energy is not limiting.
- Increase the available nutrients, because nutrients in dead phytoplankton (which are unavailable) are replaced by dissolved nutrients (which are available), while the total amount is kept constant.

The first effect dominates when energy is (nearly) limiting: all year in some lakes, only in winter in others. However, energy is present in excess of need in many lakes during significant parts of the year. In these cases, the second effect is responsible for the predicted results: yearly maxima are nutrient limited and can increase because a higher than nominal fraction of nutrients becomes available. Ultimately, of course, energy becomes limiting to the yearly maxima in all lakes, but the required retention times are much shorter than will usually be attainable.
In conclusion, flushing as a single tactic to reduce summer bloom levels is effective only in some lakes, which unfortunately are usually among the deepest and largest lakes considered and for which flushing requires enormous amounts of water. It is conceivable, however, that flushing could be more effective where relatively nutrient-poor water is available. Also, flushing could be applied in combination with other tactics, for instance to remove phosphate coming out of the bottom when an explosive flux occurs.

16.4.4. Dredging

Energy is frequently limiting in the lakes considered for PAWN, both in winter and (in some lakes) in summer. If these lakes (Randmeren, Slotermeer) were not extremely shallow, energy could perhaps become an even more important limitation. Thus the impacts of an increased depth were simulated in all presently shallow lakes.

Several important assumptions were made. First, the background extinctions were kept at their (usually high) nominal values. Because there is little information on how the background extinction relates to the physical, chemical, and biological characteristics of a lake, it was impossible to make a prediction of its future values if the depths were increased. It seems, however, that the value of $K_b$ tends to be inversely proportional to the depth (Table 8.3); thus it is not unlikely that $K_b$ would become smaller if a lake were dredged. Considering the sensitivity of the results to the value of $K_b$, it is important to design methodologies to predict its future value.

Second, the nominal nutrient concentrations have been maintained. It is conceivable that nutrient concentrations would go down initially, because dredging would automatically remove the top layer of bottom sediment, which presently can act as an important nutrient source in summer. In that case the predicted results would be upper bounds because annual maxima in the simulations are energy, rather than nutrient, limited.

The results (Fig. 16.4) indicate that dredging has a great impact on the predicted maximum bloom levels in all lakes, but particularly those in which $K_b$ is high (Slotermeer, Gooimeer). If $K_b$ were to become progressively smaller as the depth is increased and range between 1 and 2 m$^{-1}$ at a depth of 5 m, the yearly maxima would be between those computed for the Zoommeer and Wolderwijd. Obviously this still implies a considerable reduction in the potential bloom levels compared to the nominal situation.

We conclude that dredging can be highly effective, particularly if the background extinctions are to remain high. This may be the case in some locations, for instance in the Slotermeer, because in peat lakes most light is absorbed by colored, organic particles. It is
predicted that under the assumptions made the provisional chlorophyll standards could be met all year in most lakes if they were dredged to about 3 m (Slotermeer) to 7 m (Wolderwijd).

Notice that other important environmental impacts of dredging have not been investigated. For instance, dredging below a certain depth rules out any possibility for a rich, diverse ecosystem with submerged macrophytes such as existed in some of the Randmeren before phytoplankton species became dominant.

16.5. IMPACTS OF PROMISING TACTICS ON OXYGEN

The oxygen budget can be improved by tactics which reduce phytoplankton biomass (discussed in previous sections) and BOD (from wastewater). The concentration of the latter, however, is already so low in Dutch lakes that any further reduction would hardly give an improvement. This was confirmed by some tentative simulations, which are not discussed here. Thus we shall only consider tactics which reduce phytoplankton biomass concentrations.

Of all eutrophication control tactics simulated by BLOOM II, only two of the most promising have been assessed with OXYMOD: reduction of the available phosphorus concentrations and dredging. For phosphorus reduction, we have not considered the full range assessed by BLOOM II, but only reductions by 50 and by 90 percent of the current values. Other tactics were not evaluated, because of their limited success in reducing phytoplankton biomass levels. Also, we have only considered those lakes in which the oxygen problems are most serious. The Haringvliet is not considered, because it is not very susceptible to tactics; nor are the Zoommeer and the Stuwpad Lith considered, because the results would be too uncertain, for reasons explained in Sec. 13.2.

Prior to a discussion of the results of these simulations, it must be pointed out that one should be careful how to interpret the individual numbers that are computed by the model. We should consider them more as an outline of the kind of results to be expected from a tactic rather than as a detailed prediction for each individual lake. Thus conclusions will be formulated in rather general terms.

The results for the most promising tactics are shown in Table 16.2. In comparison with the nominal results of Table 13.3, the oxygen budgets are not significantly improved when available phosphorus concentrations are reduced to 50 percent of the current levels. However, almost all oxygen problems cease with a 90-percent reduction of available phosphorus: the daily average concentrations, the diurnal variations, and even the impacts of a collapsing bloom become acceptable.

Dredging to a depth of 3 m (4 m in Wolderwijd) results in a substantial reduction in the diurnal variations. The potential
duration of anaerobic conditions diminishes, too, but not to the same extent as with a P reduction of 90 percent. Dredging below 3 m does not appear particularly successful, because the improvements from a decreased phytoplankton biomass are largely canceled out by a slower reaeration, due to the greater depths. Notice that dredging could reduce available phosphorus concentrations because, P loads from the bottom might be reduced. This potential amplification of the success of dredging has not been taken into account.

Table 16.2.

IMPACTS OF PROMISING TACTICS ON OXYGEN, BASED UPON BIOMASS COMPUTATIONS BY BLOOM II

<table>
<thead>
<tr>
<th>Lake plus Tactic</th>
<th>Daily Average Oxygen Concentration</th>
<th>Absolute Maximal Oxygen Concentration</th>
<th>Maximal Diurnal Variation</th>
<th>Maximal Anaerobic Duration</th>
<th>Maximal Phytoplankton Biomass</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>ppm $O_2$</td>
<td>(d)</td>
<td>(mg C/l)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>IJsselmeer:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P red. 50%</td>
<td>8.9</td>
<td>14.1</td>
<td>8.1</td>
<td>14.5</td>
<td>1.8</td>
</tr>
<tr>
<td>P red. 90%</td>
<td>8.3</td>
<td>13.6</td>
<td>8.1</td>
<td>13.6</td>
<td>0.5</td>
</tr>
<tr>
<td>Veluwemeer:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P red. 50%</td>
<td>6.8</td>
<td>15.3</td>
<td>5.0</td>
<td>15.0</td>
<td>4.5</td>
</tr>
<tr>
<td>P red. 90%</td>
<td>6.8</td>
<td>14.4</td>
<td>6.2</td>
<td>15.0</td>
<td>1.2</td>
</tr>
<tr>
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<td>14.6</td>
<td>5.9</td>
<td>14.7</td>
<td>2.4</td>
</tr>
<tr>
<td>Depth 4 m</td>
<td>6.7</td>
<td>14.4</td>
<td>6.2</td>
<td>14.4</td>
<td>1.2</td>
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<td>Wolderwijd:</td>
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<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>P red. 50%</td>
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<td>14.6</td>
<td>6.3</td>
<td>14.8</td>
<td>3.8</td>
</tr>
<tr>
<td>P red. 90%</td>
<td>7.0</td>
<td>14.3</td>
<td>6.7</td>
<td>14.4</td>
<td>1.0</td>
</tr>
<tr>
<td>Depth 4 m</td>
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<td>14.8</td>
<td>6.5</td>
<td>15.0</td>
<td>2.5</td>
</tr>
<tr>
<td>Depth 5 m</td>
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<td>14.6</td>
<td>6.7</td>
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<td>1.5</td>
</tr>
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<td>Gooimeer:</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P red. 50%</td>
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<td>15.4</td>
<td>5.5</td>
<td>16.0</td>
<td>4.6</td>
</tr>
<tr>
<td>P red. 90%</td>
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<td>14.3</td>
<td>5.8</td>
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</tr>
<tr>
<td>Depth 3 m</td>
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<td>5.2</td>
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</tr>
<tr>
<td>Depth 4 m</td>
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<td>14.0</td>
<td>5.6</td>
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<td>Slotermeer:</td>
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</tr>
<tr>
<td>P red. 50%</td>
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<td>14.7</td>
<td>7.8</td>
<td>15.0</td>
<td>2.2</td>
</tr>
<tr>
<td>P red. 90%</td>
<td>8.1</td>
<td>14.5</td>
<td>8.0</td>
<td>14.6</td>
<td>0.8</td>
</tr>
</tbody>
</table>

NOTE: The absolute minimum and maximum oxygen concentrations are the yearly minima and maxima computed for any diurnal cycle. The maximum duration of anaerobic conditions occurs when a phytoplankton bloom suddenly collapses.

16.6. OVERVIEW OF EUTROPHICATION CONTROL TACTICS

Based upon their impacts relative to the nominal situation, the tactics discussed in the previous sections can be divided into three general categories:
Not or hardly effective.

- Silicon addition affects blooms only in a few locations during one quarter of the year; the increase in total biomass is a potential problem.

Partly effective in some locations.

- Incrementing the background extinction can be effective, particularly in lakes with a depth of more than 3 m, if sufficient colored substances are applied. Possible ecological side effects have not been investigated and are largely unknown.
- Flushing can be effective in turbid and deep lakes in which blooms are frequently energy limited, but it is unlikely that sufficient water is readily available. Flushing could have negative impacts in relatively clear or shallow lakes, because the available nutrients are increased.

Frequently effective.

- Homogeneous mixing can be an important tactic for a selective suppression of blue-green phytoplankton. The impacts on total biomass might also be significant, because presently no other species can achieve similar high blooms in winter (in all lakes) or even summer (in some lakes). In-lake experiments are recommended.
- Dredging of shallow lakes is highly effective to reduce phytoplankton bloom levels, even if the values of the background extinctions were to decrease considerably. Dredging to 3 m reduces the diurnal oxygen variations substantially, but anaerobic conditions of 6 days or less are still computed upon bloom collapse. Any further increase in depth has little impact on the oxygen conditions. However, dredging is expensive and creates a new ecosystem, which may not be desirable everywhere.
- A reduction of the phosphorus concentrations in the lakes by 40 percent or more can be effective to reduce bloom levels; the corresponding reductions in phosphorus loadings could not be computed, however. Still higher concentration reductions by 40 to 90 percent, depending on the local conditions, are necessary to reduce bloom levels below the provisional (not to say target) standard for chlorophyll. With a 50-percent reduction, the oxygen conditions hardly improve, but with a 90-percent reduction most oxygen problems cease to exist.

Future blooms were predicted from scenarios rather than (exactly) computed boundary conditions because no methodologies were available
to predict all of these. However, it is usually unrealistic to assume that a tactic can be implemented as in the most extreme simulations. For instance, it seems unlikely that phosphorus concentrations can be reduced by 90 percent, which is necessary to meet the target standard in some lakes, or that all shallow lakes would be dredged to 5 m. Furthermore, the existing regional and seasonal differences between the lakes considered for PAWN (Chap. 9) have great implications for the simulated impacts of control tactics: the lakes respond quite differently to similar changes in boundary conditions. Summarizing all results, we may reach two important conclusions:

- Implementation of a lake's most effective tactic will, by itself, frequently be insufficient to meet the provisional biomass standard.
- No tactic we have considered is effective in all lakes.

Thus an overall improvement of eutrophication is possible only with a regional approach. National policies such as phosphorus removal do not seem sufficiently effective to prevent considerable local violations of the biomass standards (see also Sec. 16.7).

16.7. SOME ALTERNATIVE RECOMMENDATIONS TO NATIONAL POLICIES

Currently, phosphate removal is considered to be the main policy in the Netherlands to combat eutrophication. Additional or alternative measures, however, are not excluded. Implementation of the phosphate removal policy is based on regional phosphate balances, including contributions of border-crossing rivers such as the Rijn. In addition to this approach, we propose a lake-specific approach. First, the function(s) of a particular body of water should be determined: recreation, drinking water, nature reserve, etc. Next, local standards should be obtained from these functions for biomass, visibility, pollutants, etc. These standards might or might not be the same as national standards. The final step would be to find tactics tailored to the local situation to reduce violations of the local standards for bloom sizes, limiting factors, nutrient loadings, and dominant species, and to combine these tactics into a regional sanitation policy.

As pointed out previously, such a policy will usually involve several different but related tactics. Although we have not investigated combinations of tactics for PAWN, our results enable us to make some suggestions for logical combinations, although it is by no means our intention to be complete. These suggestions serve only as possible examples.

One possibility could be to reduce phosphate loadings significantly, but additionally remove the top few decimeters of the bottom, which presently contains large deposits of nutrients. If the results are
insufficient because still too many nutrients are released by the sediments, one possible suggestion would be to flush the lake during critical periods, for instance when an explosive flux is taking place. Of course this kind of flushing can be effective only if the loadings are permanently reduced.

Other combinations of tactics could aim at a reduction of available energy. Thus, a moderate increase in depth could be combined with artificial mixing, for example in storage reservoirs, which have to be dredged anyway. To be effective, however, the depth would not have to be as great as in the Biesbosch reservoirs, especially not if the background extinction had a moderate value.

NOTES

1. This is the provisional phosphate standard specified in the IMP of 1975-1979. The new IMP of 1980-1984, which was not available for this study, lowers the provisional limit for phosphate to 0.2 mg/l.

2. This range reflects differences in specific extinction coefficients (Table 8.2) and dry weight to chlorophyll ratios.

3. Blue-green phytoplankton can obtain upward or downward velocities by changing the size of their gas vacuoles; dinoflagellates can make active swimming movements. Both can obtain velocities on the order of meters per day.

4. Inhomogeneous mixing should not be confused with (temperature) stratifications, although for instance Microcystis can indeed form thin layered scums. Also it is important that species with active buoyancy control can even increase their net production rates, if on the average they seem to be homogeneously distributed, because they can react to variations in the environmental conditions and "be at the right place at the right time."

5. It seemed unlikely that homogeneous mixing would be used as a control tactic in these two waters.

6. Low total biomass values, usually less than 50 mg chlorophyll per m^3, mostly of diatoms and flagellates.

7. If we had assumed an increase in the nutrient concentrations, the increase in biomass would have been even more pronounced.

REFERENCES

