EUTROPHICATION OF WATERS

MONITORING, ASSESSMENT AND CONTROL

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CHAIRMAN'S PREFACE (1)

The following report is a synthesis of the main results of the OECD Cooperative Programme on Eutrophication. It is the outcome of several years' concerted effort by 18 Member countries. The objectives were to establish, through international cooperation, a basis for eutrophication control of inland waters (lakes and reservoirs in particular), and to develop better guidelines for fixing nutrient load criteria compatible with water use objectives.

The present report is both complementary and supplementary to the four Regional Project Reports (2) already published. In parallel with the OECD study programme, progress has been made in other areas, particularly in dynamic modelling. The results of the OECD study and approach have already been successfully applied in several instances in North America, Europe and elsewhere. It can be anticipated that -- while knowledge of eutrophication and its control methods are advancing -- the OECD results presented here will continue to provide a basic reference in eutrophication control studies.

STATUTE OF THE REPORT

The conclusion of this report have been successively agreed by the Water Management Policy Group, the Environment Committee and finally the Council. The technical part of this synthesis report has also been approved by the Water Management Policy Group.

1. Dr. Richard Vollenweider, Chairman of the Technical Bureau.
2. See Appendix 1 of this report.

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SUMMARY AND CONCLUSIONS

The OECD Cooperative Programme on Eutrophication (1) was designed to quantify the relationship between nutrient load in waters and their trophic reaction.

The lengthy programme, on which 18 Member countries and numerous institutes and researchers have been cooperating on a voluntary basis, has achieved its objectives through conducting extensive and complex studies which make use of all available recent information.

The programme was carried out by establishing four main regional projects covering a wide variety of geographic and limnological situations (Alpine, Nordic, Reservoir and Shallow Lakes, and the North American Project). Four regional Project Reports have been completed, and a fifth report has been developed which tests the OECD results on bodies of waters not included in the analytical part.

The objectives of the programme have been largely achieved, and application of the results to practical control of eutrophication is possible. However, it is recommended that the results be handled with caution and not applied to cases which lie outside the ranges and situations covered by the programme.

The main control strategy is reduction of the external load. In cases where such a reduction to the required tolerance level is impracticable, or impossible (as e.g. in situations of intensive use of the catchment system), management measures other than nutrient reduction have to be employed. Such alternative measures, however, can only be defined using advanced modelling techniques. Notwithstanding the positive achivements of the programme, eutrophication remains a very complex problem and numerous questions are still unanswered.

Man-made accelerated eutrophication of inland waters in OECD Member countries can generally be viewed as an undesirable degradation of the environment resulting in a deterioration of water quality which interferes with most of the beneficial uses of waters; it is causing, in many cases, significant economic losses. Although nitrogen or some other factor may in some particular instances outweigh the role of phosphorus as the limiting factor, most attention here is based on phosphorus control; however, where the term "nutrients" appears, both phosphorus and nitrogen may be considered in order to include potential cases of nitrogen limitation.

A. WATER RESOURCE PLANNING

The results of the OECD programme have shown that:

a) in most cases, phosphorus is the factor which determines the development of eutrophication;

1. Cooperative Programme on Monitoring of Inland Waters, Eutrophication Control.

b) even when another nutrient such as nitrogen is (occasionally or normally) the limiting factor, phosphorus may still be made to play the role of limiting factor through appropriate control.

It is safer and generally more economic to take early preventive measures to control eutrophication than to develop curative strategies later when water quality has already deteriorated. Member countries should initiate and maintain policies and programmes to prevent and counteract the undesirable symptoms of eutrophication in water bodies as a fundamental part of Water Resource Planning.

An objective assessment of the significance of eutrophication in Member countries' water resources should be made in relation to the particular usage, purpose or function allocated to the different types of water bodies.

This should include:

- an assessment of the extent and the importance of the problem, present and future;
- an assessment of the main sources of nutrients potentially causing eutrophication (both point and diffuse sources);
- an assessment of measures taken (e.g. nutrient countrol facilities already installed) as well as an assessment of their operational efficiency;
- an assessment of both direct and indirect costs.

Control of point sources of pollution from municipalities and industries is usually given priority as it is generally the most cost-effective measure. Appropriate technologies for phosphorus reduction (or nitrogen reduction, where appropriate) are available, and the precipitation process is the most effective method to eliminate a high proportion of phosphorus from controlled effluents.

After reduction of phosphorus from point sources, the relative role of phosphorus from diffuse sources will increase. This means that measures against diffuse sources may become necessary if improvement of water quality cannot be achieved by further elimination of phosphorus point sources.

Diffuse source control is more difficult to achieve. Yet, in many cases, effective prevention of eutrophication, or restoration of eutrophied waters cannot be achieved without such control.

Therefore, the improvement of *all* aspects of agricultural practices which contribute nutrients to water bodies should be encouraged, with special reference to:

- control of waste from intensive animal husbandry;
- control of the dose, period and methods of fertiliser application in order to achieve minimal loss and optimum up-take by crops;
- control of erosion and run-off from tillage land and from forestry operations;
- control of over-irrigation (fertiliser leaching to ground waters) (1).

Attention should also be given to urban run-off and storm overflows, which normally by-pass conventional treatment plants. A more integrated approach may be necessary, and should include modifications to system design standards, source controls, sewer separation, flow detention, and overflow treatment.

In certain situations nutrient contributions from septic tanks may be important. Potential contributions from such sources should be carefully assessed, although no generally applicable methodology exists to control septic tank leaching. However, when septic tank effluent is dispersed by tiled drains in a field, a high degree of phosphorus removal can be achieved for a considerable time, especially with certain types of soils.

Although nutrient load control is the only truly effective long-term strategy against eutrophication, it is evident that in certain situations, effective nutrient control

1. See *Impact of Fertilisers and Agricultural Waste Products on the Quality of Wastes*, OECD 1973.

is not possible. Various technologically feasible options, applied in the water body itself, are available for such situations, the choice of which depends upon the particular limnological conditions of the water body in question. Examples of the various options are referred to in Chapter 10 of the Synthesis Report. Eutrophication cannot really be controlled by herbicides and algicides. As the use of these products presents a hazard, they should only be considered as an interim measure to alleviate symptoms in exceptional cases and strictly limited environments.

Although the OECD Programme was designed to control the mainly negative effects of eutrophication, there may be exceptional cases where an increase in the trophic level can have limited beneficial aspects (e.g. for increasing the population of certain fish species). The potential benefits are, however, rapidly outweighed by the negative effects on all uses as trophic levels increase.

Greater attention should be given to the economic use and the promotion of by-products from water pollution and eutrophication control measures. Instances include: low cost energy gains from sludge digestion (bio-gas); potential use of sludge as fertiliser products; irrigation with waste water ("fertirrigation"); harvesting of primary products from controlled lagooning etc.

In order to aid the understanding of eutrophication control and its cost-effectiveness aspects, well-documented case histories should be available to experts in appropriate administrations in Member countries; the socio-economic aspects should be well documented so that improved public support can be gained.

Steps should be taken to ensure that adequate specialist advice is available for the relevant expertise in countries. It is particularly felt that while some countries can call on highly qualified limnologists and engineers others cannot.

B. TECHNICAL AND SCIENTIFIC BACKGROUND

Monitoring

It is essential, for the control of eutrophication, that an adequate study is initiated in order to plan the measures necessary to improve water quality to the standard required for its intended use. Experience has shown that in many cases a comprehensive monitoring programme of adequate duration will be necessary.

Close cooperation between limnologists, hydrologists, agriculturalists, sanitary engineers, and involved authorities is essential in any programme so that maximum efficiency and economy of the study is achieved.

The OECD list of agreed parameters and methods of measurements (Tables 3.3 and 3.4 of the Synthesis Report) together with the other recommended guidelines, should be taken as initial reference. Any new methodologies emerging over recent years should also be considered.

The frequency of sampling, both spacial and temporal, is an important but expensive component in a study programme. No universally valid sampling scheme can be given, but the following principles should be taken into account int relation to local conditions.

- In lakes: the spacial and temporal frequency of sampling has to be commensurate with the spacial and temporal variability. In lakes with high storage capacity, sampling once a month per station and depth selected should provide an adequate picture of the average conditions; in some cases, and for some parameters, once every three months may be sufficient. On the other hand, lakes exhibiting high spacial and temporal variability, as well as lakes with an irregular flushing regime, may require more frequent sampling.

In order to evaluate long-term changes, the number of years required to cover a Study Programme has to be commensurate with the hydrologic variability over time. Long-term hydrological information must be taken as a reference basis.

The same principles apply to station selection and frequency of sampling on tributaries, and their basins. In addition, station selection will have to depend on land-use characteristics, present and future. It is usually impractical to monitor *all* nutrient sources that contribute to the total nutrient load; surveillance priority should go to those sources which make a large contribution to the total load. Sources that are known to supply nutrients at a relatively even rate can be sampled less frequently, while sources with a changing load regime require a higher frequency of sampling, particularly during periods of peak discharge.

Nutrient Load Assessment

Assessment of the total nutrient load is a crucial problem which demands greater attention, particularly with regard to non-point (diffuse) sources. Results from the experience gained in the OECD Study Programme indicates that the likelihood of errors in load estimates can be as high as ± 50%. This emphasises the importance that must be attached to the planning of a comprehensive measurement programme of adequate duration. This should locate the seasonal and other variations of nutrient input and reduce the level of potential error in loading estimates to a minimum, both as an annual load and the load at particularly sensitive periods.

For practical purposes, the following break-down scheme is recommended for estimating the total load.

a) external:
- the phosphorus and nitrogen load via the tributaries (including "point" sources along the tributaries, and "diffuse" or "non-point" sources on the drainage basin);
- the point and diffuse sources load which directly enter the lake through the shores;
- the phosphorus and nitrogen load which falls on to the surface of a lake as wet or dry precipitation;

b) internal:
- phosphorus and nitrogen which re-enter from sediments. The net result of the interchange of nutrients between water and bottom sediments can be estimated by making nutrient balances covering relevant periods of time.

Measurements of concentration (both in point and diffuse sources where applicable) must always be combined with measurements or appropriate estimates of flow. In tributaries, gauging stations need to be installed at key river transects, if the already existing stations are not sufficient or adequate.

To a limited extent, load from diffuse sources may be estimated from export coefficients; however, this must be done with utmost caution, and only after the applicability on the selected co-efficient to the local conditions to be surveilled, has been established. Point-sources contribution can be estimated from known source strength co-efficients. However, it is evident that these indirect estimates cannot be taken as a substitute for direct measurements.

Particular attention is drawn to combined storm and sanitary sewer overflows which discharge untreated sewage to watercourses in times of excess rainfall. This aspect is specially significant where nutrient reduction processes are in operation at sewage treatment plants, since stormwater overflows provide a by-pass for all treatment. Moreover, regular storm sewer discharge (from separated systems), which is generally untreated, must also be assessed for its relative nutrient input.

Assessment of the nutrient loading resulting form precipitation is important. It is desirable to liaise with meteorological and other departments in order to obtain better estimates of the contribution by precipitation, both wet and dry.

Trophic Assessment

The proper assessment of the limiting nutrient parameter is crucial to the success of a control programme. Although the OECD study has shown that in most cases phosphorus is the controlling factor, a prerequisite of any case study is a careful evaluation of possible exceptions. For this, more emphasis should be placed on bio-assay techniques.

Among the recommended OECD parameters for trophic assessment, particular emphasis should be directed at dynamic parameters such as hypolimnetic oxygen depletion and primary production. More attention should also be given to identification and enumeration of phytoplankton, and to the interrelationship between phyto- and zooplankton. Changes in algal species composition with shifting trophic conditions cannot accurately be predicted. However, in many management applications such prediction can be helpful (e.g. with regard to taste, odour and filtration problems in drinking water supply). The littoral regime, in relation to macrophytes and attached algae (e.g. cladophora), should not be disregarded as such organisms create nuisance conditions.

Loading Criteria and Nutrient Reduction Objectives

The improved and modified "phosphorus load/lake response" relationship, which has been developed from the Cooperative Programme, should be used as a valuable tool for assessing the expected effects of phosphorus reduction in a water body and for establishing the phosphorus loading/water quality criteria for the different types of water bodies, according to their intended usage. However, management should be aware of the uncertainties built into the prediction formulae (described in the Synthesis Report) and evaluate these uncertainties relative to their objectives.

It has to be recognised that the results of the OECD Programme may not be directly applicable to geographic regions and limnological situations which differ greatly in their characteristics from those covered by the Programme.

Lakes of complex morphometry and embayments of large lakes should be considered as separate entities when applying the empirical "nutrient load/trophic response" relationship formulae.

If information regarding the most limiting nutrient cannot be obtained, the "phosphorus load/lake response" relationship should not be applied to lakes with a phosphorus concentration higher than 100 or 200 mg/m^3 (depending on the characteristics of the lake); however, in exceptional cases, for very shallow lakes, the upper limit may be up to 300 mg/m^3.

Until the relationship between nutrient load and dynamic parameters is better understood, water quality objectives for the intended water use should be set at the appropriate levels, using Chlorophyll *a* concentration as a measure of phytoplankton biomass. For most water uses, it will be necessary to consider mean Chlorophyll *a* and peak Chlorophyll *a*.

Criteria for nutrient reduction should be set according to the ecological function and to the use intended for a particular water body (e.g. drinking water, recreation, multiple use, etc.).

When *all* external (point and diffuse) sources of phosphorus and their relative contribution to the total load (including projections for future growth) are identified and assessed, priorities for nutrient load control measures regarding the responsible sources should be established. This should take into consideration the importance of the various sources and the assessed effectiveness of their control in order to achieve the desired level of water quality.

During the planning stage of reservoirs and man-made lakes, it is recommended that an assessment be made of the expected trophic state of the planned lake. If this indicates that the lake is likely to become permanently eutrophic, provision should be made for the introduction of appropriate control measures, which can be based primarily on nutrient control, or on other principles such as light limitation by artificial mixing (deep reservoirs).

C. MANAGEMENT CONSIDERATIONS

The impact of eutrophication on drinking water supply may be serious. For a number of reasons it generally reduces its final quality and may diminish its safety. Furthermore, it makes its preparation more difficult and costly.

The problems encountered include: rapid clogging of filters by diatoms and other algae; disturbance of floculation treatment by organic substances; persistent and unpleasant taste and odour (e.g. geosmine); abnormal concentration of substances such as manganese, iron and ammonia giving rise to colour or other disturbances; risk of increased bacterial growth in drinking water due to the fouling of the distribution networks and the nutrient content.

Furthermore, because of the high content of organic substances in eutrophied waters and some of the problems listed above (taste, ammonia, regrowth of organisms etc.) these waters are often extensively chlorinated during treatment as well as in the initial transportation and final distribution networks. High levels of both chlorine and organic substances lead to significant concentrations of organochlorinated compounds in drinking water, and these substances are now considered to be potentially hazardous for human health (carcinogenic risk) (1). Waters for potable use should thus be protected from eutrophication.

Many development programmes throughout the world, whether for agricultural, industrial or urban develoment, are based on extensive water resource projects which involve high investments. The reservoirs and channels created are prone to rapid development of eutrophication which may handicap entire projects.

Careful attention should be paid, from the early planning stages, to preventing eutrophication, and where the spread of the phenomenon seems uncontrollable, other strategies (for instance the use of storage capacity of aquifers) should be considered, especially where potable water supplies are involved.

The impact of eutrophication on recreation and tourism is probably the most sensitive area for the public. It may severely alter the recreational value of many water bodies and impair related activities (swimming, fishing, etc.) as a result of the objectionable aspect of the waters, such as reduced transparency, odour, and increased incidence of stinging insects, swimmer's itch, etc. Both social impacts and economic losses may be important and make eutrophication control necessary.

Adequate treatment of urban sewage does not exist in all OECD countries, and only a small proportion of all waste waters receive "tertiary treatment". Installation of adequate treatment facilities should be encouraged, and, where eutrophication is a major problem, should be combined with phosphorus removal facilities. However, the socio-economic aspect of alternative schemes for controlling the various types of phosphorus source responsible for eutrophication, needs careful evaluation.

In most countries detergent products contain a high proportion of polyphosphates in their formulae. A few countries have found it necessary to reduce the polyphosphate content in detergents as an effective step towards decreasing the nutrient load in waters. However, this is not a substitute for establishment of waste

1. OECD report on Control of Organochlorinated Compounds in Drinking Waters.

water treatment facilities. Detergent products as a whole account for about half the phosphorus present in urban waste waters. Furthermore, as polyphosphates are in a very soluble form and thus readily available for plant growth, their impact on nutrient load and eutrophication remains particularly important. The development of technically, environmentally and economically satisfactory alternatives to polyphosphates in detergents still continues to be an important and not yet entirely resolved issue.

When incorrectly managed, considerable quantities of organic wastes (i.e. animal manures, abattoir slurries, sewage treatment sludges, etc.) are annually washed off to waters, or discharged to refuse dumps which leach to underground or surface waters and are responsible for severe pollution and nutrient release. Once properly prepared and standardised for their fertilising value, they could be used instead of chemical fertilisers, provided they can compete with these fertilisers which are easier to apply, relatively cheap (due to favourable fiscal conditions) and backed by efficient commercial organisations. In line with OECD Member countries' policies for recycling, the direct re-use by agriculture or the marketing of these organic fertilisers should be encouraged by adequate regulatory and economic instruments.

D. RESEARCH NEEDS

The results of the OECD Cooperative Programme (as summarised in the various regional reports and the Synthesis Report) have provided ample evidence that efficient and economically feasible control of eutrophication is possible right now. However, this does not mean that research has come to a close. Indeed, the study has brought to light the existence of many gaps and uncertainties in our knowledge and there is a need for further efforts in the development of new methodologies, procedures and conceptual approaches.

The following is but a cursory list of areas identified where research should be promoted.

1. Methods

Improved methods and standardisation are required in the following areas:

a) More reliable biomass parameters to reduce some of the uncertainties connected with chlorophyll measurements;
b) Standardisation of primary production measurements;
c) Unification of procedures to calcultate hypolimnetic oxygen depletion rates and assessment of hypolimnetic redox conditions;
d) Improvement on loading estimates, with particular regard to diffuse source estimates and establishment of reliable export coefficients for different land-use categories under varied climatic conditions;
e) Improved estimates of internal loading vis-à-vis external loading;
f) Methodologies to identify "available" forms vis-à-vis total forms of nutrients.

2. Nutrient load/trophic level relationship

a) Improving the present methodology, to include nitrogen together with phosphorus, and dynamic parameters such as hypolimnetic redox conditions and primary production together with chlorophyll.
b) Establishing corresponding relationships relative to available forms of nitrogen;
c) Establishing the modifying effect of various physical and chemical regimes such as high flushing situations, stratification and mixing cycles, ionic composition, pH, humic substance content, etc.;

d) Role of internal loading;

e) Developing a basis for establishing load/trophic relationships for climatic regions and aquatic environments (e.g. marine) not covered by the Programme.

3. **Biotic interactions**

a) Role of macrophytes and attached littoral algae vis-à-vis phytoplankton communities under varied trophic conditions;

b) Successional patterns in aquatic phyto-biota as related to changes in trophic conditions with particular emphasis on bloom-forming species, and species creating other problems such as taste and odour, clogging of filters, etc.;

c) Importance of food-chain interactions relating to trophic conditions.

4. **Effects**

a) Establishing the extent to which eutrophied waters contribute, with chlorination treatment, to organochlorine formation in drinking water, due to the high level of organic precursors and the more intensive chlorination they receive at various stages of treatment and transportation;

b) Establishing the interrelationship between eutrophication and metal and other toxicities resulting from pollution load;

c) Encouraging studies on the impact of eutrophication on human health, especially with regard to the spread of waterborne diseases in temperate and trophical climates;

d) Pursuing studies on the beneficial and detrimental effects of eutrophication on freshwater and coastal marine fisheries.

5. **Technological**

a) Improvement of techniques and methods to make use of the sludge derived from wastewater treatment plants (particularly when there are special phosphorus removal techniques) as a valuable fertiliser, in view of the large amount produced and the cost of its disposal;

b) Promotion of treatment procedures incorporating land disposal techniques and by-product utilisation;

c) Development of a fully acceptable alternative for polyphosphates in detergent products.

6. **Managerial**

a) Review of agreed "List of Standard Parameters and Methods of Measurement" methodologies with a view to reducing the time and cost of sampling, analysis and interpretation necessary to predict trophic response and advise on restoration strategies;

b) Promotion of multivariate dynamic modelling techniques for the assessment of trophic response and alternative control strategies;

c) Re-examination of the field of control strategies with a view to reducing costs and developing effective and inexpensive methods and technologies;

d) Identification and demonstration of emerging biological nutrient removal techniques with and without chemical assistance;

e) Identification, development and demonstration of nutrient removal, technologies which focus on the available forms of nutrients;

f) Development of low cost/high removal efficiency, low energy intensive/high by-product recovery and re-use technologies.

INTRODUCTION

THE PROBLEM OF EUTROPHICATION

Early in the 1960s, it became obvious that a large number of lakes and reservoirs, particularly those located in industrialised countries, were rapidly changing in character and becoming increasingly fertile (eutrophication) because of the addition of plant nutrients originating largely from human activities. The main nutrient sources were municipal and industrial wastewater and agricultural and urban runoffs.

Eutrophication, which may be natural ou "man-made", is the response in water to overenrichment by nutrients, particularly phosphorus and nitrogen. "Man-made" eutrophication, in the absence of control measures, proceeds much faster than the natural phenomenon and is one of the major types of water pollution.

The resultant increase in fertility in affected lakes, reservoirs, slow-flowing rivers and certain coastal waters causes symptoms such as algal blooms, heavy growth of certain rooted aquatic plants, algal mats, deoxygenation and, in some cases, unpleasant tastes and odours. These often adversaly affect the vital uses of water such as water supply, fisheries, or recreation and impair aesthetic qualities. In short, man-made eutrophication of inland bodies of water has become synonymous with the deterioration of water quality and is frequently the cause of considerable cost increases.

Algal blooms often clog filters of municipal and industrial water supplies necessitating frequent and costly cleaning procedures. The shores of formerly clean lakes develop algal slimes, excessive algal turbidity or dense growth of certain rooted aquatic plants and filamentous algae in shallow areas. As a result, lakes become unattractive for bathing, boating and other water-oriented recreations and result in severe economic loss to established resorts. The former users of such lakes are forced to travel elsewhere at added expense. Fish production often increases but the species composition changes. Economically important species such as trout decline or disappear and are often replaced by coarser fish or lower economic value.

Man-made accelerated eutrophication can be reversed by the elimination or reduction of the nutrient supply from sources such as municipal and industrial wastewaters, agricultural wastes and fertilisers. In most cases, however, it is not possible to eliminate all sources of nutrient supply. It is important, therefore, to understand the quantitative relationship between nutrient supply and the degree of eutrophication (i.e. trophic response, manifesting itself in plankton biomass). From that sound lake management strategy may be developed in order to obtain the desired water quality by reducing the amount of nutrient supplies to eutrophic waters at minimum cost. To reach this goal, the Organisation for Economic Cooperation and Development (OECD) organised an international programme with the aim of developing a sound data base for evaluating the response of inland bodies of water to various rates of nutrient supply over a large geographical area which, after analysis, would lead to the development of scientific lake management principles for the control of excessive eutrophication of water.

THE RATIONALE AND THE BACKGROUND OF THE OECD EUTROPHICATION PROGRAMME

In 1967 a group of experts under the Chairmanship of the late Professor O. Jaag (EAWAG, Zürich) recommended to the OECD that a comprehensive survey be made of the existing literature on eutrophication processes. This led to the publication of a report, "Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication" by Vollenweider (1968). This Report introduced the concept of relationship between nutrient loading and lake response but also stressed the inadequacy of limnological data for broad generalisations and for producing precise guidelines for eutrophication control.

In 1968 the practical experience already gained by Member countries in this water management problem was discussed at a symposium on "Eutrophication in Large Lakes and Impoundments" in Uppsala, Sweden; a report of this was published by OECD in 1970.

In spite of considerable advances in eutrophication control, many basic questions concerning eutrophication still remained unanswered and it became obvious that a broader limnological data base was required for inter-comparison between bodies of water and for the assessment of the status of lake eutrophication. The nutrient loading concept and the related concept of loading tolerance had been consolidated and accepted by a large segment of the international scientific community, but there were many who still maintained that carbon and other growth factors rather than phosphorus or nitrogen limit algal growth in many lakes.

In 1971, the OECD established a Steering Group on Eutrophication. In 1973, an "Agreed Programme on Evaluation of Eutrophication Control" was approved and adopted and the Steering Group on Eutrophication Control was charged with the responsibility for developing and coordinating the agreed programme, taking into account its effectiveness, cost and feasibility. Four *ad hoc* expert groups carried out the programme:

1. *Expert Group on Detergents* (published 1974);
2. *Expert Group on Impact of Fertilisers and Agricultural Waste Products on the Quality of Waters* (published 1973);
3. *Expert Group on Waste Water Treatment Processes for Phosphorus and Nitrogen Removal* (published 1973);
4. *Planning Group on Measurements and Monitoring* (published 1973).

The three expert groups and the planning group completed their reports in 1972. The planning report, published in 1973, "Summary Report of the Agreed Monitoring Projects on Eutrophication of Waters" (OECD, 1973) gave a common system of agreed measurements, guidelines on background data and comments on existing methods and sampling. It also outlined the basis for an international programme of measurements and monitoring of waters being undertaken by interested OECD Member countries.

1. OBJECTIVES OF THE PROGRAMME

1.1. SCIENTIFIC BASIS

Scientifically speaking, eutrophication is only a special aspect on normal lake productivity. Seen in this perspective, studies on eutrophication, both on lakes and reservoirs, have to follow the same conceptual references as lake studies in general. Lake productivity is, on the one hand, the expression of the physico-chemical complexes of the catchment system in which a given lake is located, and on the other hand, it is a function of the internal physicochemical and biological dynamics of the component compartments of the lake itself.

In order to facilitate reading of the report in the light of its principal conceptual references, the two productivity determining aspects are illustrated in Figures 1.1 and 1.2.

Figure 1.1 emphasises the general dependency of lake productivity on the properties of the catchment system in terms of geophysical and geochemical factors (level 1) which, in turn, determine the specific properties of the waters drained (level 2).

Under unaltered pristine conditions, the only major transfer compartment between the two levels would be the natural vegetation-soil complex. Throughout man's history, this complex has undergone the most radical alterations in terms of land use, urbanisation and industrialisation. The accelerated rate at which these alterations have occurred over recent decades is the main factor responsible for alterations in the natural productivity of lakes.

Transfer from level 1 to level 2 is commonly measured in terms of source strength, export coefficients, deflux coefficients, etc., and transfers from level 2 to level 3, as regards nutrients, are measured in terms of loading of the receiving water body.

Figure 1.2 depicts the main internal dynamics of a lake system, and how this relates to external (L_E) and internal (L_I) nutrient loading. It further emphasises the principal pathways of the external loading components which may reach the nutrient pool in the form of dissolved inorganics and hence become immediately biologically available, or as organic compounds made available only after appropriate microbiological break-down, or in the form of detritus. Productivity is then measured by the size and/or velocity of replacement of one or more of the main compartments in terms of biomass, primary production, oxygen consumption, etc.

The relative importance of these various related and interrelated compartments and components depends ultimately on the specific geographic, geo-morphological, climatological, morphometric, hydrodynamic, and other circumstances. The large potential variation in these components has to be taken into account in specific studies, and it has been one of the most serious stumbling blocks for comparative studies in the past.

In recognition of these aspects, the OECD Cooperative Programme has been designed to overcome some (yet by no means all) of the related problems. A degree of simplifications has been unavoidable in many respects at the comparative level,

Figure 1.1

THE THREE LEVELS DETERMINING THE PRODUCTIVITY OF BODIES OF WATER

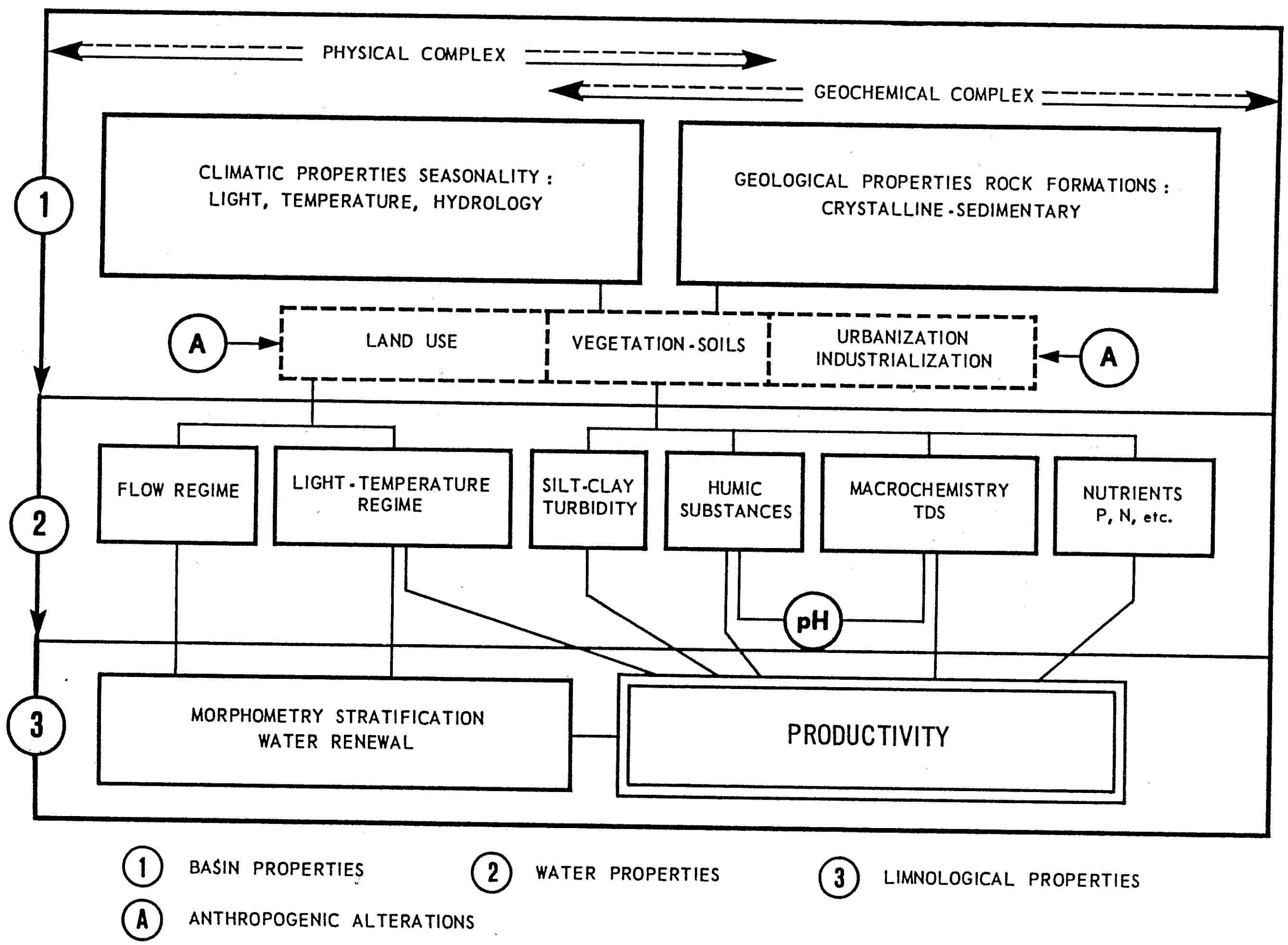
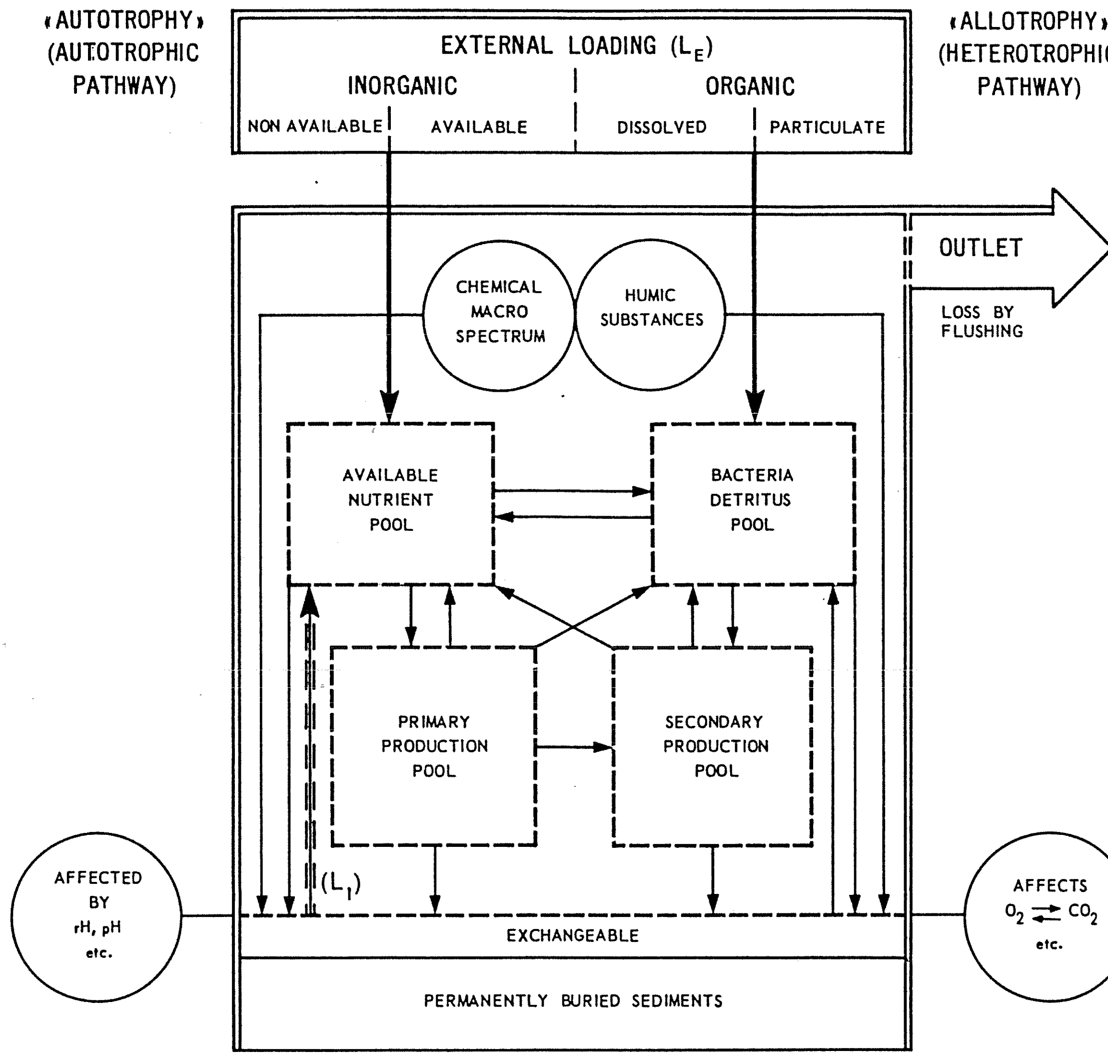


Figure 1.2



«AUTOTROPHY»
(AUTOTROPHIC PATHWAY)
«ALLOTROPHY»
(HETEROTROPHIC PATHWAY)
EXTERNAL LOADING (L_E)
INORGANIC
ORGANIC
NON AVAILABLE
AVAILABLE
DISSOLVED
PARTICULATE
OUTLET
LOSS BY FLUSHING
CHEMICAL MACRO SPECTRUM
HUMIC SUBSTANCES
AVAILABLE NUTRIENT POOL
BACTERIA DETRITUS POOL
PRIMARY PRODUCTION POOL
SECONDARY PRODUCTION POOL
(L_I)
AFFECTED BY rH, pH etc.
AFFECTS $O_2 \rightleftharpoons CO_2$ etc.
EXCHANGEABLE
PERMANENTLY BURIED SEDIMENTS

though not necessarily at the individual study level. This short introduction aims to clarify this aspect and also to indicate that practical intervention into the eutrophication process is possible at various levels, though in the programme attention has been directed mainly towards clarifying the role of external nutrient loading, and its implication for lake management.

1.2. MAIN OBJECTIVES

Central to the OECD Cooperative Programme on Eutrophication is the concept that accelerated eutrophication is caused by excessive nutrient load (externally and internally), and that there exists a quantifiable relationship between nutrient load and trophic reaction of the affected body(ies) of water: a relationship which depends on a variety of physiographic factors such as morphometry, hydrology, general climatology, etc.

Accordingly, the study programme was designed to elucidate these dependencies, and in particular, was intended to:

a) lead to a better understanding of the relationship, expressed in quantitative terms, between nutrient supply to and trophic response of lakes;
b) determine which one of the principal nutritional factors, nitrogen or phosphorus, is the main factor causing eutrophication;
c) elucidate the role of modifying conditions and factors other than nutrients, such as morphometric, hydrologic, physical and chemical conditions, involved in determining the ultimate trophic response in a given body of water.

1.3. SUBSIDING QUESTIONS

In order to achieve the main objectives, the study programme considered a number of subsidiary questions summarised as follows:

a) Is it possible to quantify the trophic response, in terms of biomass and related dynamics, to the nutritional conditions observed in lakes ?
b) Is it possible to quantify the relationship between nutrient supply to lakes and observed nutritional conditions of these lakes?
c) Is it possible to relate the trophic response of lakes directly to nutrient supply?
d) Is it possible to provide explanation for non-conforming lake situations?
e) Is it possible to determine, for practical management, criteria relating trophic categories to loading conditions, and vice versa?

The outcome of the programme has largely answered the questions posed and these will be presented in the appropriate context in the following chapters. Particular aspects which have not been entirely resolved are identified.

The programme, as a whole, can be described as highly successful, and the results are immediately applicable to practical management of lake and reservoir eutrophication. However, it must be stressed that this is true only for those classes of water bodies covered by this programme (see § 3.3.).

2. PROGRAMME ORGANISATION AND LIMNOLOGICAL SITUATIONS COVERED

For both organisational convenience and regional characteristics of the water-bodies studied, the programme has been broken down into four projects (Alpine, Nordic, Reservoirs and Shallow Lakes, and North American). The main characteristics and the rationale for these projects are summarised in Table 2.1

It is important to note here that, in geographic terms, the *study is restricted mainly to lakes of the temperate zone*, including some border line cases. Except for a few isolated cases, marine environments have not been considered; also, very special limnological environments have not been examined. The lakes, for which data have been used in one way or another in this study, are listed in Table 2.2.

Regional Summary Reports were prepared and these form the basis for this Synthesis Report (Appendix 1).

The OECD study has been directed by a Technical Bureau composed of the regional project leaders and consultants (Table 2.1) under the chairmanship of Dr. R. A. Vollenweider. Additional members of the Technical Bureau (Dr.H.Golterman, Dr. J. Kerekes, Ing. C. Milway, Mr. G. Dorin) have provided extensive assistance and advice throughout the programme.

Table 2.1

CHARACTERISATION OF THE FOUR REGIONAL PROJECTS
OF THE OECD COOPERATIVE PROGRAMME FOR MONITORING OF INLAND WATERS

1. Alpine Project

Project Leaders: M. H. Ambühl, H. Löffler, and O. Ravera.
Consultant: Hj. Fricker.

In the Alpine regions are the headwaters of a large number of European waters. The Alpine lakes are of social, economic and scientific significance because they represent a great natural amenity, an important centre for tourism, and are the subject of a large number of significant limnological studies. Their ecology is characterised by an abundant variety of species, historically well documented, which are vulnerable to man's intervention. The Alpine zones represent similar hydrological conditions due to comparable geography, geology, and ecology, and they share certain river basins and commissions. Thirty lakes, consisting of 38 lake basins, are included in this project.

2. Nordic Project

Project Leader: F. C. Forsberg
Consultants: S.O. Ryding, J. Lønholdt

The Nordic Project includes the cool climatic zone of the Baltic and North Sea areas, i.e. the lakes resulting from the retreat of the great quaternary glaciers. Reasonably comparable conditions exist throughout this project area as well as similar levels of economic development and pollution, with close political, cultural and scientific links. Many lakes are used for water supply and recreation and are the subject of detailed limnological investigations. Ten lakes, consisting of 15 lake basins, are included in this project.

3. Reservoir and Shallow Lakes Project

Project Leader: H. Bernhardt
Consultant: J. Clasen

This project includes man-made lakes and reservoirs and other relatively shallow lakes, lagoons and estuarine waters covering a wide variety of geographical situations. They have great economic and social value (e.g. water supply, water sports, fishing). The rationale for lumping these waterbodies into a separate project is that water quality control by manipulation of hydrological or other factors is usually more feasible for and applicable to these waterbodies. Thirty-two waterbodies are included in this project.

4. North American Project

Project Leaders: N. Jaworski, T. Maloney and R.A. Vollenweider
Consultants: G.F. Lee, W. Rast

In constrast to the European projects, this project is not restricted to a narrow geographic area or to specific waterbody types. It includes waterbodies in various regions of Canada and the United States covering a broad spectrum of trophic and morphometric conditions and ranging from ultra-oligotrophic pristine lakes to highly eutrophic ones, and from small, shallow, highly flushed lakes to the Laurentian Great Lakes and three sections of the Potomac Estuary.
Whereas the U.S. lakes have been treated according to standard recommendations, the Canadian lakes selected are used as a test-case to study the feasibility of applying the OECD results to a large variety of limnological situations.

Table 2.2.

THE WATERBODIES YEARS OF INVESTIGATION AND THEIR TROPHIC STATUS IN THE OECD COOPERATIVE PROGRAMME
(Trophic status symbols are explained at the end of Table)

No	Trophic status	Waterbody	No of years investigated	Dates
		NORDIC PROJECT		
101	E	Esrøm	3	73-75
102	E	Mossø 1	3	73-75
103	E	Mossø 2	3	73-75
104	E	Gjersjøen	4	72-75
105	M	Mjøsa	4	73-76
106	E	Boren	4	73-76
107	E	Mälaren	3	73-75
108	O	Vättern	1	74
109	E	Päijanne 1	3	73-75
110	O	Päijanne 2	2	76-76

No	Trophic status	Waterbody	No of years investigated	Dates
111	M	Päijanne 3	2	73-75
112	O	Päijanne 4	1	73
113	O	Päijanne 5	2	73-75
114	O	Pääjärvi	7	70-76
115	E	Tuusulanjärvi	3	74-76
		ALPINE PROJECT		
201	O	Lunzer Untersee	2	75-76
202	O	Feldsee	2	74-75
203	E	Bodensee Obersee	1	
204	M	Lago di Mergozzo	1	75
		Lago di Lugano		
205	E	Lago di Ponte Tresa	1	
206	E	Lago di Capolago	1	
207	E	Lago di Melide	1	
208	E	Lago di Gandria	1	
209	E	Lago di Lugano	1	
210	E	Lago di Agno	1	
211	E	Lago di Morcote	1	
212	E	Lago di Figino	1	
213	E	Piburger See	3	74-76
214	M	Ossiacher See	3	74-76
215	O	Attersee	2	75-76
216	E	Greifensee	2	75-76
217	M	Walensee	3	73-75
218	E	Baldeggersee	1	74
219	E	Hallwilersee	1	73
220	E	Sempachersee	2	75-76
		Vierwaldstättersee		
221	M	Kreuztrichter	2	73-76
222	M	Gersau	1	73
223	M	Urnersee	1	73
224	M	Lago Maggiore	2	72-73
225	M	Walensee	1	76
226	E	Zürich Untersee	1	
227	E	Zürich Obersee	1	
228	M	Lac Léman	1	76
229	E	Lago di Oggiono	1	73
230	E	Lago di Pusiano	1	73
231	M	Lago di Montorfano	1	
232	M	Lago di Annone	1	73
233	E	Lago del Segrino	1	73
234	E	Lago d'Alserio	1	73
235	E	Lac d'Annecy	1	
236	E	Lac Nantua	1	
237	O	Lac Tazenat	1	
238	M	Lac Pavin	1	
239	E	Lac Aydat	1	

No	Trophic status	Waterbody	No of years investigated	Dates
		SHALLOW LAKES AND RESERVOIRS		
301	E	Wahnbach	3	74-76
302	O	Olef	2	75-76
303	E	Sorpe	1	74
304	E	Möhne	2	73-75
305	O	Verse	1	76
306	O	Söse	3	74-76
307	E	Ennepe	2	74-75
308	O	Fürwigge	1	76
309	E	De Grote Rug	3	74-76
310	E	Briellse Meer	2	74-75
311	E	Petrusplaat	3	74-76
312	E	Hondred en Dertig	3	74-76
313	E	Braakman II	3	74-76
314	E	Braakman III	3	74-76
315	O	Vechten	1	74
316	E	Tjeukemeer	1	75
317	E	Queen Elizabeth II	1	69
318	E	Farmoor	3	71-73
319	E	Grafham Water	2	69-73
320	E	Lough Neagh	1	74
321	E	Loch Leven	1	74
322	M	Lough Leane	2	73-74
323	E	Lough Ennel	2	75-76
324	E	Biwa South	3	74-76
325	O	Biwa North	3	74-76
326	E	Kasumigaura West	3	74-76
327	E	Kasumigaura North	3	74-76
328	M	Mount Bold	3	73-76
329	M	Prospect	2	75-76
330	M	El Burguillo	3	74-76
331	E	Nisramont	3	74-76
332	O	Eupen	3	74-76
		U.S.A. PROJECT		
401	E	Blackhawk	1	72-73
402	E	Brownie	1	72
403	E	Calhoun	1	71
404	E	Camelot-Sherwood	1	72-73
405	E	Canadarago	2	68-69
406	M	Cayuga	2	72-73
407	E	Cedar	1	71
408	E	Cox Hollow	1	72-73
409	O	Dogfish	2	71-72
411	E	Dutch Hollow	1	72-73
412	O	George	1	72-73
413	E	Harriet	1	71
414	E	Isles	1	71

No	Trophic status	Waterbody	No of years investigated	Dates
416	E-M	Kerr Roanoke	1	75
417	E-M	Kerr Nutbush	1	75
419	O	Lamb	2	71-72
421	O	Meander	2	71-72
422	E	Mendota	2	65-66
423	O	Michigan Open Water	2	71-74
425	E	Lower Minnetonka, 1969	1	69
426	E	Lower Minnetonka 1973	1	73
428	H	Potomac Upper Reach	5	66-70
429	H	Potomac Middle Reach	5	66-70
430	E	Potomac Lower Reach	5	66-70
431	E	Redstone	1	72-73
432	E	Sallie	2	72-73
433	M	Sammamish	1	
434	E	Shagawa	1	72
435	E	Stewart	1	72-73
436	O	Tahoe	1	73-74
441	E	East Twin	4	71-74
443	E	West Twin	4	71-74
446	E	Twin Valley	1	72-73
447	E	Virginia	1	72-73
448	UO	Waldo	1	74
449	E	Washington 1957	1	57
450	E	Washington 1964	1	64
451	M	Washington 1971	1	71
453	M	Weir	1	74-75
454	E	Wingra	1	70-71

* As indicated by the investigator:

UO = ultra-oligotrophic
O = oligotrophic
M = mesotrophic
E = eutrophic
H = hypertrophic

3. DATA BASE

3.1. BACKGROUND DATA

A substantial effort was made to obtain the background data necessary for each water body studied, to understand its general limnological characteristics and those aspects which are important for its management. The principal items requested for this purpose are listed in Table 3.1. Specific information is available from the individual contributors, and has been in part summarised in the Regional Project Reports.

3.2. MEASUREMENT DATA

Measurement data requested for the programme have been subdivided into *essential* and *supplementary* data, and are listed in Table 3.2. The first category refers to those measurements deemed absolutely necessary for any subsequent statistical analysis, whereas the second category refers to data which each individual contributor deemed necessary to understand more completely his particular case. Accordingly, the choice of these latter has been left with the individual investigator. The suggested set of measurement data is listed in the form of measurement categories in Table 3.3, and in the form of specific reporting categories in Table 3.4.

In order to avoid misunderstanding of the *modus operandi* of the programme, it should be stressed that the definition of "essential" does not imply a mandatory request for all measurement categories listed to be carried out. However, only those case studies for which a sufficient number of essential data measurements were available have been included in the OECD Programme. In practice this meant that specialised, limnological studies, incomplete in terms of the OECD Programme, could not be considered as "OECD study material", regardless of their scientific importance.

By the same token, this *modus operandi* explains part of the variation in data points used for different correlation analyses. Although the limited data set used in some of the correlation calculations, may reduce their level of significance, the study shows that the essential conclusions from the total analysis remain valid for most of the principal relationship considered. Insufficient resolution for others will be mentioned in appropriate sections.

Table 3.1

RECOMMENDED BACKGROUND DATA (GEOGRAPHIC, MORPHOMETRIC AND HYDROLOGIC), FOR MEASURING AND MONITORING EUTROPHICATION IN INLAND WATERS

A. Geographic

1. Latitude and longitude (of the centroid of the water area).
2. Altitude of the water above sea level (or below).
3. Catchment area (including the area of surface water).
4. General climatic data (ice coverage, average monthly air temperatures, wind pattern, evaporation and evapotranspiration, etc.).
5. General geological characteristics, particular nature of bedrock, subsoil and soils, and the importance of land erosion. Description of type of lake.
6. Vegetation.
7. Population.
8. Lande usage (industrial, urban, agricultural, etc.).
9. Use of water (drinking, sport, fishing, etc.).
10. Sewage and effluent discharge (population and industry).

B. Morphometric and hydrologic

1. Surface area of water - length, width (maximum and average), shore length, etc.
2. Volume of water (with information on regulation).
3. Maximum and average depth (by basins if applicable).
4. Location of exceptional depths and the surface area ratio of deep to shallow waters.
5. Ratio of epi- over hypolimnion.
6. Duration of stratification.
7. Nature of lake sediments.
8. Seasonal variation of monthly precipitation together with maximum and minimum conditions on drainage basin.
9. Inflow and outflow of water (also underground).
10. Water currents.
11. Water renewal time (retention time).

C. Ecologic

1. Bacteria.
2. Phytoplankton.
3. Algal bioassays.
4. Bottom flora.
5. Macrophytes.
6. Bottom fauna.
7. Zooplankton.
8. Fish.

Table 3.2

SUMMARY OF THE LIMNOLOGICAL VARIABLES
CONSIDERED IN THE OECD COOPERATIVE PROJECTS
FOR THE MONITORING OF EUTROPHICATION OF INLAND WATERS
(after OECD, 1973b)

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1. physical

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

2. Chemical

- *Essential:* pH; dissolved O_2; phosphorus, nitrogen; SiO_2; alkalinity and acidity; Ca^{++} & Mg^{++}; Na^+ & K^+; $SO_4^=$ & Cl^-; Total Fe;
- *Desirable:* Other trace elements (e.g. Manganese, Molybdenum) and micropollutants (pesticides, etc.); H_2S, CH_4.

3. Biological

- *Essential:* Phytoplankton (chlorophyll *a*); oxygen production (primary production), organic carbon.
- *Desirable:* Phytoplankton identification by genera and counting; C^{14} uptake; zooplankton identification and counting.

a) The *essential* are those variables which are deemed necessary to provide an adequate knowledge of trophic conditions, scope for comparison between waterbodies and improvement in the ability to predict change whether as an improvement in water quality, as a result of the introduction of control measures or by deterioration as a consequence of their absence.

b) The *desirable* are those variables which are appropriate for laboratories with larger resources, or that desire to pursue some areas of investigation to a greater extent. This type of measurement supplements the essential data.

Table 3.3

CATEGORISATION OF LIMNOLOGICAL VARIABLES
FOR MEASURING AND MONITORING EUTROPHICATION IN INLAND WATERS

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1. Resultant Variables

a) Short-term variability: high
- Phytoplankton biomass
- Major algal groups and dominants species
- Chlorophyll *a* and other phytopigments
- Particulate organic carbon and nitrogen
- Daily primary production rates
- Secchi disc transparency

b) Short-term variability: moderate to low
- Zooplankton standing stock
- Bottom fauna standing stock
- Epilimnetic ΔP, ΔN, ΔSi (Δ - difference between winter and summer concentrations)
- Hypolimnetic O_2 and ΔO_2; H_2S, CH_4 etc:
- Annual primary production.

2. Causative Variables

- Measured nutrient (P and N) loadings
- Phosphorus concentrations (total; soluble reactive)
- Nitrogen concentrations (total Kjeldahl; $NO_3 + NH_3$)
- Reactive silica
- Others (e.g. micro elements).

3. Related Descriptive Variables

- Temperature
- Conductivity, pH, alkalinity, major ions
- Insolation and optical properties of waters
- Others (colour, inorganic turbidity, etc.).

Table 3.4

CONDENSED MEASUREMENT DATA USED FOR DATA ELABORATION
IN THE OECD PROGRAMME FOR MEASURING EUTROPHICATION OF INLAND WATERS
(Alpine, Nordic, Shallow Lakes and Reservoirs Projects)

Name of Lake: Year:

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1. Nutrient Load and Output Units: tons/year

Variables:
total-P
PO_4-P
total-N
$(NH_4-N) + (NO_3-N) + (NO_2-N)$

2. Nutrient Concentrations Units: mg/m^3

Variables:	Reporting units for each variable:
total-P	– spring overturn peak values of mixed layer
PO_4-P	– mean spring overturn concentration of total lake
tota-N	
$(NO_3-N)+(NO_2-N)$	– mean annual concentration of total lake;
NH_4-N	– mean concentration of euphotic zone, winter
SiO_2	– mean concentration of euphotic zone, spring
	– mean concentration of euphotic zone, summer
	– mean concentration of euphotic zone, autumn
	– mean concentration of euphotic zone, year

3. Planktonic Primary Production

Variables:	Units:
annual production	$g\ C/m^2/year$
minimal daily production	$g\ C/m^2/day$
maximum daily production	$g\ C/m^2/day$
average daily production, winter	$g\ C/m^2/day$
average daily production, spring	$g\ C/m^2/day$
average daily production, summer	$g\ C/m^2/day$
average daily production, autumn	$g\ C/m^2/day$
highest yearly maximum on depth profiles	$g\ C/m^3/day$
lowest yearly maximum on depth profiles	$g\ C/m^3/day$

4. **Depth of Euphotic Zone** Units:

Method of calculation:

- from production profiles
- from photometric measurements
- from Secchi disc readings

Reporting Units:

- maximum value
- minimum value
- mean value

5. **Phytoplankton and Chlorophyll a of the Euphotic Zone**

Depth of euphotic zone is given in metres.

Concentration and standing mass

Variables:	Units: Concentration	Standing mass
— chlorophyll *a*	mg/m^3	mg/m^2
- algal volume	mg/m^3	mg/m^2

Reporting units for each variable:

- annual mean concentration and standing mass
- annual maximum concentration and standing mass
- mean winter concentration and standing mass
- mean spring concentration and standing mass
- mean summer concentration and standing mass
- mean autumn concentration and standing mass

6. **Secchi Depth**

Reporting units (in metres).

- annual mean
- mean winter
- mean spring
- mean summer
- mean autumn
- annual maximum
- annual minimum

7. **Hypolimnetic Oxygen Depletion** Units: $g\ O_2/m^2$, $g\ O_2/m^3$

Reporting units:

— Difference between onset and end of summer stratification	$g\ O_2/m^2$
(unit volume of hypolimnion)	$g\ O_2/m^3$
— Average montly depletion rate	$g\ O_2/m^2/month$
(unit volume of hypolimnion)	$g\ O_2/m^3/month$

8. **Physical and Hydrological Characteristics of Lake Catchment Area**

Variable:	Units:
– catchment area excluding lake	km^2
- lake surface area	km^2
— volume of lake	m^3

- yearly water input . . . . m^3
- yearly water output. . . . . m^3
- area of hypolimnion . . . . m^2
- volume of hypolimnion. . . . . m^3
- maximum depth . . . . m

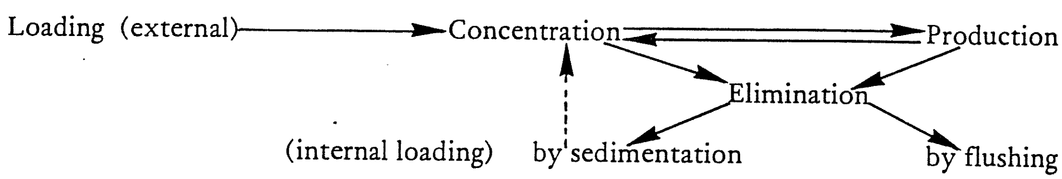
3.3. RANGE AND VARIATION OF THE DATA BASE

The programme covered in total some 150 lakes, ranging from "pond-size" lakes to the Great North American Lakes, the latter representing about 10% of the total freshwater resources of the world. The programme has included a substantial number of freshwater bodies, although as mentioned before, not all specific freshwater types have been adequately covered.

Geometric mean values and absolute range of variation of all essential parameters for which a sufficient data base exists, are listed in Table 3.5. This table includes all except the Canadian data which are, in fact, within the ranges listed.

3.4. DATA ELABORATION

In considering the scope and intent of the programme, and anticipating the complexity and problems connected with the study, it was decided at the very beginning to elaborate the data primarily with statistical techniques, and to present the synthesis data in the form of correlations and regressions. A simple conceptual mass balance model, which relates directly to the questions posed in § 1.3 (a type of one-box model), was used to guide the data integration and analysis. This model can be summarised as follows:



From this calculation, models have been derived, and their parameters have been determined by iteration techniques. The theoretical framework for the approach taken is presented in Appendix 2.

More complex dynamic models were deliberately excluded for the purposes of this programme. The rationale for this is as follows:

a) the OECD programme, as a whole, could not provide the data base required for such models;
b) dynamic models are in a development phase, and;
c) they do not provide an appropriate basis for large-scale comparative purposes, as envisaged in the OECD study.

It is understood, however, that statistical procedures and dynamic modelling are not mutually exclusive. Some of the specific studies have been conducted in such a way that dynamic techniques can be applied; their pursuit is left to the individual contributor.

Table 3.5

GEOMETRIC MEANS AND EXTREMES AND THE NUMBER OF OBSERVATIONS OF SELECTED VARIABLES USED FOR CORRELATION AND REGRESSION ANALYSIS IN THE OECD EUTROPHICATION PROGRAMME

Variable	Symbol	Units	Geometric Min.	Geometric Mean	Geometric Max.	N
Lake area	A_0	km^2	.025	6.6	58,000	126
Mean depth	$\bar{z}$	m	1.7	14.3	313	126
Hypolimnion area	Ha	km^2	.06	6.9	535.1	31
Water residence time	T(w)	y	.016	1.2	700	112
Flushing rate	r(w)	1/y	.001	.81	63.7	112
Hydraulic load	q_a	m/y	.447	10.7	447.6	112
Loading of total phosphorus	L(P)	$g/m^2/y$	.017	1.2	80	102
Loading of PO_4	L(PO_4)	$g/m^2/y$	.010	1.3	106	42
Loading of total nitrogen	L(N)	$g/m^2/y$	.810	27.5	1710	56
Loading of inorganic nitrogen	L(IN)	$g/m^2/y$	.330	19.1	1606	62
Outflow of total phosphorus	O(P)	$g/m^2/y$	.040	.74	80.0	61
Outflow of PO_4-P	O(PO_4–P)	$g/m^2/y$	.010	.34	96.1	53
Outflow of total nitrogen	O(N)	$g/m^2/y$	.200	19.7	1710	48
Outflow of inorganic nitrogen	O(IN)	$g/m^2/y$	.100	14.5	1284	50
Annual mean concentration of total phosphorus	$[\overline{P}]_\lambda$	mg/m^3	3.00	47.1	750	115
Spring concentration of total phosphorus	$[\overline{P}]_\lambda^{sp}$	mg/m3	5.6	44.2	1120	68
Annual mean euphotic total phosphorus	$[\overline{P}]_e$	mg/m^3	3.8	40.1	373	77
Winter euphotic total phosphorus	$[\overline{P}]_e^w$	mg/m^3	3.6	45.5	445	80
Spring euphotic total phosphorus	$[\overline{P}]_e^{sp}$	mg/m^3	3.7	42.2	410	64
Summer euphotic total phosphorrus	$[\overline{P}]_e^s$	mg/m^3	2.0	42.0	750	105
Fall euphotic total phosphorus	$[\overline{P}]_e^f$	mg/m^3	4.0	38.6	487	65
Annual mean inflow of total phosphorus	$[\overline{P}]_j$	mg/m^3	4.7	112.2	1425	92
Predicted total phosphorus from loading	$\frac{[\overline{P}]_j}{1+\sqrt{T(w)}}$	mg/m^3	1.8	49.3	663	92
Mean annual outflow of total phosphorus	$[\overline{P}]_0$	mg/m^3	3.7	40.6	555	64
Mean annual inorganic phosphorus	$[\overline{PO_4-P}]_\lambda$	mg/m^3	0.2	16.1	891	99
Mean annual total nitrogen	$[\overline{N}]_\lambda$	mg/m^3	263.	1244	6095	58
Mean annual inorganic nitrogen	$[\overline{IN}]_\lambda$	mg/m^3	10.0	500	5395	97
Mean annual euphotic chlorophyll *a*	$[\overline{chl}]$	mg/m^3	0.3	8.4	89	96
Mean winter euphotic chlorophyll *a*	$[\overline{chl}]^w$	mg/m^3	0.4	1.6	17.5	14
Mean spring euphotic chlorophyll *a*	$[\overline{chl}]^{sp}$	mg/m^3	1.3	7.9	105	37
Mean summer euphotic chlorophyll *a*	$[\overline{chl}]^s$	mg/m^3	0.2	9.2	107	58
Mean fall euphotic chlorophyll *a*	$[\overline{chl}]^f$	mg/m^3	0.3	3.7	26.7	36
Annual peak chlorophyll *a*	max chl	mg/m^3	2.0	22.8	275	61
Annual mean euphotic chlorophyll *a* annual mean total phosphorus ratio	$[\overline{chl}] : [\overline{P}]_\lambda$	ratio	.027	.27	.95	78

Variable	Symbol	Units	Geometric Min.	Geometric Mean	Geometric Max.	N
Annual peak euphotic chlorophyll *a* annual mean total phosphorus ratio	$\frac{\text{max}}{\text{chl}} : \overline{[P]}_\lambda$	ratio	.08	.58	3.8	52
Mean annual Secchi disc transparency	$\overline{[Sec]}^y$	1/m	0.8	3.3	28.3	94
Mean summer Secchi disc transparency	$\overline{[Sec]}^s$	1/m	0.5	3.13	28.0	52
Annual primary production	ΣPP	g $C/m^2/y$	5.9	188	1166	67
Residence time of total phosphorus	T(P)	y	.006	.53	24.5	97
Settling velocity of phosphorus	$V_s(P)$	m/y	.66	15.8	143	87

3.5. SIGNIFICANCE OF THE RESULTS

In accordance with the philosophy adopted for data integration and elaboration, it has to be stressed that the end product of the various elaborations has to be visualised as a *description of statistical behaviour of a selected population of lakes relative to the factors analysed, rather than as a description of the behaviour of a single lake.*

To evaluate the results of the OECD study correctly, it is important to keep this differentiation in mind. From the statistically derived relationships can be discerned the degree of accordance (or discordance) in the behavioural pattern of a family of lakes against which a simple situation can be tested. Outliers from such patterns are of particular interest, implying that their basic behaviour is not fully understood (1).

Generally speaking, what has been achieved through the OECD study lies, says, half way between the historic position indivually according to which each lake has to be individually understood and an advanced level of insight – but not yet attained – which would make it possible to deduce the reaction of waterbodies with a high degree of precision from a few parameters.

Despiste this limitation, the outcome of the OECD programme represents a considerable step in the desired direction. *Its results can confidently be applied to practical management of lakes on the condition that managers are sufficiently aware of the limitations of the study itself, and the variety of limnological situations and conditions to which the results are applied (see § 9).*

3.6. SOME IDENTIFIED PROBLEM AREAS REGARDING PRIMARY AND SYNTHESIS DATA

Whereas the scattering of data, observed in all the correlation analyses performed, in many cases reflect real variability, in others cases (often difficult to identify) the scattering is due to uncertainties in the basic data provided. Uncertainties of this kind, arise partly from analytical errors, partly from the kind of calculation procedures used,

1. From a management point of view, these are the cases where funds should be allocated for further studies rather than to those which are within the general pattern.

and partly from the specific calculation models proposed in the study. A clear distinction of the relative importance of each of the listed conditions is difficult, and for most cases impossible.

Inlake Nutrient Data. In those projects (Alpine, Northern) where it was possible to conduct appropriate ring tests during the study, the analytical uncertainties have been minimised, though not entirely eliminated. In the U.S.A. project most of the investigators conducted analyses of standard samples as part of the work required by their supporting agencies. In the Reservoir project analytical uncertainties relating to data from different laboratories are unquestionably more serious. Logistic difficulties (e.g. distances involved, modus operandi, etc.) made it virtually impossible to cope adequately with this problem.

Nutrient Loadings. Three sources of uncertainty besides analytical errors, contribute to the overall uncertainty of loading data: *a)* sampling design; *b)* load calculation; *c)* year to year variability. Large differences in sampling design in the different single studies were unavoidable, as were differences in the method of calculating the total nutrient load which in many cases was partially or even wholly done from indirect estimates (e.g. direct discharges from sewer systems). Further experience showed that in some cases the year to year variability is substantial, depending on annual meteorological variability, but this can only be recognised in long-term studies. The study period for most OECD lakes did not exceed a two to three year cycle and in many cases information for only one year was available. Further, it should be recognised that nutrient load calculated per year fails to show how the load is distributed throughout the year. Under certain hydrological conditions, 50% of the yearly load can reach a body of water in less than six months. Similarly, seasonal operations, such as summer resorts, food processing plants, may deliver a substantial portion of the annual load during a short period.

These different factors contribute to the overall uncertainty of a reported loading value. The reliability of such values is in no case estimated to be better than ∓ 35%, and hence must be a main source of data scattering.

Biomass. "Biomass", in its various components, is the principal response characteristic of the tropho-dynamic situation of a lake. However, with regard to analytical measurement techniques, its hypothetical nature is not always recognised. In other words, there is no single and easily measured parameter available to characterise quantitatively "biomass".

For practical purposes, *chlorophyll* has been chosen as the standard parameter for phytoplankton biomass. The related analytical problems are well known, and even more important, chlorophyll per unit of phytoplankton biomass may vary both from season to season, and between lakes depending on, for example, the algal composition of phytoplankton.

Apart from this inherent problem, the values reported throughout the programme are somewhat uneven. The reported average chlorophyll values are normally yearly averages, yet in some cases refer to growing season averages. Where such cases could be identified, they have been excluded from the final correlation analyses.

Tropho-dynamic Interrelationships. More complete analyses of tropho-dynamic interrelationships, and their bearing on eutrophication manifestations, (e.g. the interrelation between phyto- and zooplankton) have not been considered in the OECD study as a standard requirement.

However, in a substantial number of cases, measurements of primary productivity have been carried out and from these yearly primary production has been estimated. Further more, hypolimnetic oxygen depletion rates have been estimated, but it was impossible to arrive at a common methodology for the respective calculations.

Every effort has been made to obtain reliable information on the hydrologic regime of the lakes studied. Several gaps in our understanding of the importance of seasonal and long-term variability do exist. Estimation of the hydrological regime has been particularly difficult in some of the reservoir studies and multi-basin lakes.

With regard to seasonal epilimnion/hypolimnion differentiation, efforts have been made in the Alpine Project to improve on certain correlations, by considering this aspect. Dynamic modelling techniques have been used in three lakes of the Alpine programme to evaluate alternative management options. In most cases the lakes have been treated as "mixed reactors" (Appendix 2). More detailed analyses of the importance of seasonal variation in terms of mixing conditions, thermal stratification, ice coverage, etc., on lake productivity have still to be evaluated.

Complex morphometric situations (differentiation into distinct basins, embayments, etc.) have been considered where appropriate. However, difficulties persist for handling even the relatively simple situation of this type, often encountered in reservoirs with a high length to width ratio. In such cases, definition of "average conditions" is somewhat arbitrary because at times considerable variations exist along the main axes of such lakes. It should also be noted that in reservoirs, the main station used for comparison, was often located near the water outlet, a fact which may have contributed to the results of regression analyses deviating somewhat from those of other projects. It should be noted that the examples are not clear cut even within the reservoir project, thus an unequivocal assessment of the influence of the morphometric characteristics on the results cannot be made.

In addition to reservoirs, difficulties can be encountered in the estimation of the average inlake concentration (representative for the whole lake) in lakes where large horizontal heterogeneities exist. Such concentrations could be seriously overestimated if the sampling station is located near to an important point source nutrient inflow. Inadequate estimation of the average inlake concentration in some cases undoubtedly contributed to the scattering of the data points in the analyses (see § 6.4.1.1.).

Concluding Remarks

This discussion of the shortcomings and limitations of the data and their analyses is not intended to give the impression that the results and conclusions presented are only of limited value. These problems have been discussed in order to focus attention on the complexity of limnological research which, in the past, has made valid comparison between bodies of water almost impossible. In contrast the OECD study has shown that substantial progress in comparative lake studies has been made despite seemingly insurmontable difficulties. With this preamble in mind, the following chapters have been kept straightforward, concentrating on the overall results, rather than on single cases i.e. the various single deviations and uncertainties are not all explained.

4. CORRELATIONS BETWEEN TROPHIC INDICATORS AND NUTRITIONAL CONDITIONS IN LAKES PRODUCTION-LIMITING FACTORS

Preamble. For the scope of this report, the results obtained from correlation analyses are discussed separately from the results presented later regarding regressions. As the two aspects are interrelated, they are usually considered jointly. The rationale for this separation derives from the intention to make clear:

a) that not all significant correlations found necessarily imply a functional interrelationship; and

b) that in the case of regression analyses, attention is given to the predictive aspect whereby questions always arise about which of the possible regression equations is most meaningful for this purpose.

4.1 NITROGEN VERSUS PHOSPHORUS

Though there is no major reason to presume that, under pristine conditions, these two factors should be correlated, there exists in the OECD study lakes a general tendency for both nitrogen and phosphorus to increase in concentration from oligotrophy to eutrophy ($r = .75$, $P > 0.99$) (Figure 4.1). The reason for this apparent association probably stems from the fact that in culturally eutrophied waters, the supply of both phosphorus and nitrogen increase in parallel, although the principal source of each substance may be different.

The tendency for nitrogen and phosphorus to increase in parallel makes it difficult to determine the relative importance of the two factors in the eutrophication process. Accordingly, it is impossible to speculate solely on the basis of nutrient conditions found in a lake, which one of the two factors is limiting production. This question can only be resolved by careful analysis of all pertinent information.

It should be noted however:

a) that relative to the phosphorus concentrations, the nitrogen/phosphorus ratio, on average, decreases from more than 100 on the oligotrophic side to less than 10 on the eutrophic side. This can be interpreted as a tendency for lakes to shift from phosphorus dependency to nitrogen dependency with increasing trophy;

b) that specific lakes may deviate from this rule, independent of their trophic characteristics. Hypertrophic lakes, for example, may not be nutrient-controlled at all but light-limited instead. Deep mixing and high water replenishment rates may also reduce the effect of nutritional conditions on production.

4.2 ORTHO-PHOSPHATE-P VERSUS TOTAL-PHOSPHORUS: MINERAL NITROGEN VERSUS TOTAL NITROGEN

Both the ortho-phosphate/total phosphorus relationship (Figure 4.2) and the mineral nitrogen/total nitrogen relationship (figure 4.3) differ from the relationship discussed in § 4.1 in the sense that, with increasing trophy, the mineral component tends to become the dominant fraction. On average, the orthophosphate-P fraction increases from less than 20% for Total-P concentrations of 10 mg/m^3 and less, to over 45% for Total-P concentrations of 200 mg/m^3 and over. Though the trend is similar, the fractional increase in mineral nitrogen relative to Total-N is less dramatic, increasing from 60% at Total-N concentrations of 500 mg/m^3 and less, to 70% at 5000 mg/m^3 and over.

For both cases, individual lakes may be at variance with the rule.

The importance of these trends is that, with increasing trophy a correspondingly higher fraction of easily metabolised mineral components become available for algal growth. In part, this greater availability depends on the metabolic processes taking place in the lake, but it is also due to a correspondingly greater supply of these components in culturally eutrophied lakes, i.e. depending on the sources causing eutrophication.

In management terms, these findings mean that the control of sources of nitrogen and phosphorus of high biological availability (dissolved mineral fractions) probably have a greater effect on the reversal of eutrophication than an unselective control, though equal in relative terms, of all sources. In other words, priority is to be given to control of biologically readily available components.

4.3. MINERAL NITROGEN VERSUS ORTHO-PHOSPHATE-P; PRELIMINARY ASSESSMENT OF PREVAILING NUTRIENT LIMITATION

Although a definitive assessment of the primary limiting factor cannot be made without due consideration of the biological component, some preliminary, yet pertinent, statements may be made here.

It is generally assumed that the mineral components (forms of inorganic nitrogen, orthophosphates) control *growth rates* (not necessarily algal biomass). A preliminary judgement on whether nitrogen or phosphorus is the growth regulating factor can be made from considering the mineral nitrogen/orthophosphate-P ratio. Complementary techniques to this are bioassay studies.

The relationship between mineral nitrogen and orthophosphate found in the OECD lakes (annual averages), is summarised in Figure 4.4. Assuming that lakes with an inorganic N/ortho-P ratio of more than 15 are not growth-limited by nitrogen, then about 70% of the cases studied would fall into the category of phosphorus-limited lakes; if one further includes lakes having ratios between 15 and 7, then 85% would fall into this category. Accordingly, the majority of lakes studied appear to be controlled by phosphorus. This conclusion will be further substantial below (see § 4.4).

An attempt to assess the growth rate dependency from the N/P ratio was made in the Nordic Project using bioassays in addition to standard analyses. From a total of 238 bioassays carried out in several lakes, 22 cases were found to be a typically N-limited with Total-N/Total-P ratios < 34, whereas the remaining 216 assays showed typical P-limitation with Total-N/Total-P ratios > 13. By employing the median values and 90% variation, only 5% of the N-limited samples were above the Total-N/Total-P ratio of 36, and 5% of the P-limited samples were found below a ratio of 13; a similar distribution was found if cases were related to the inorganic N/ortho-P ratio.

Figure 4.1

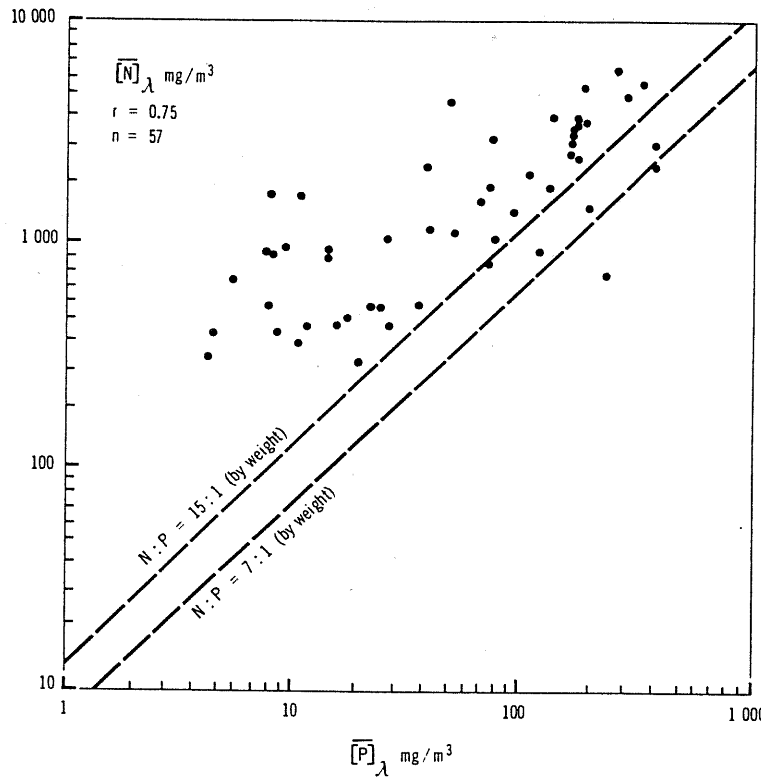


Figure 4.2

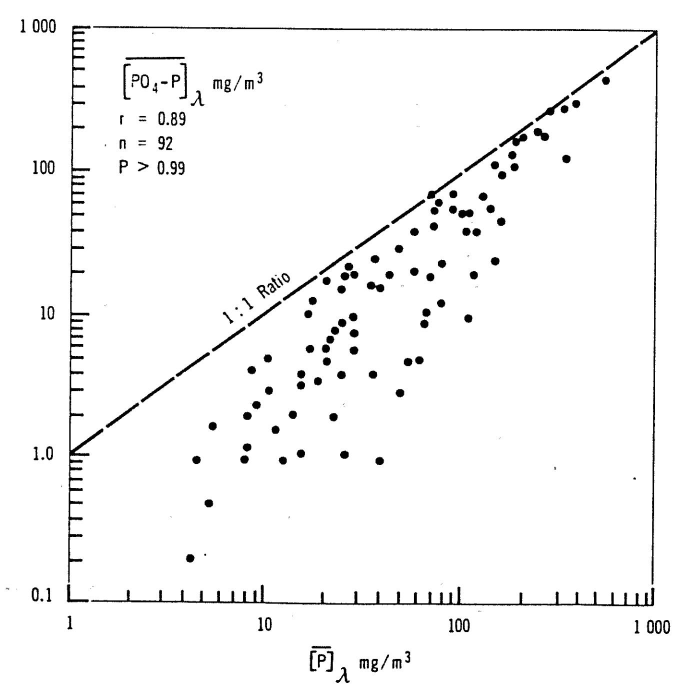


Figure 4.3

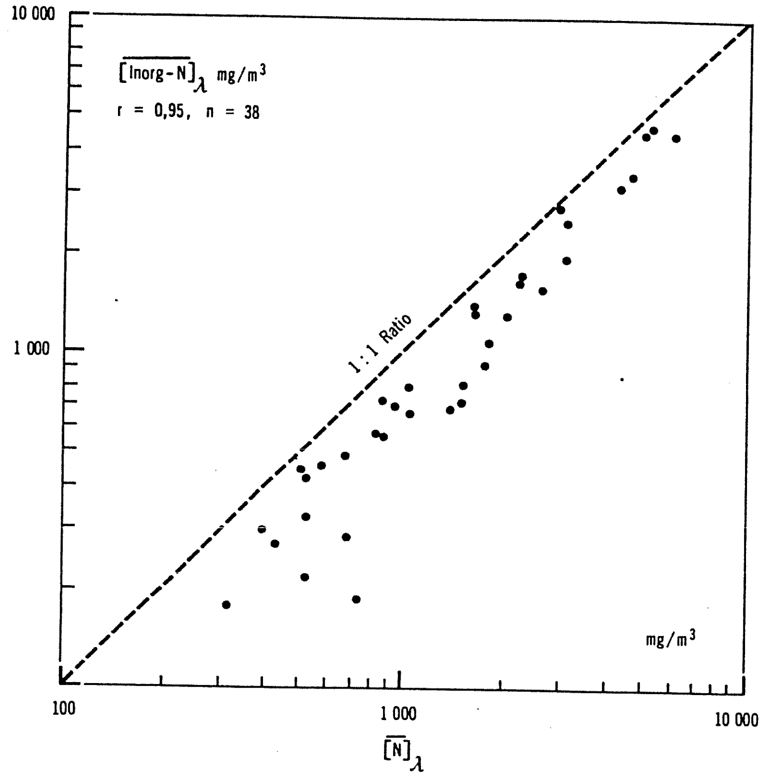
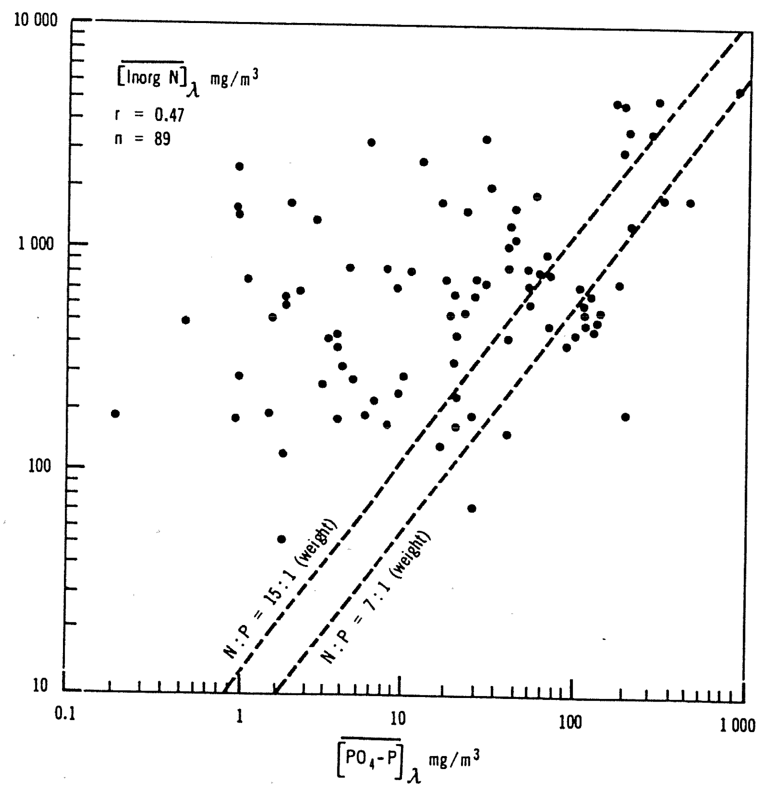


Figure 4.4



The seasonal variation of nutrient limitation was assessed further in the Nordic Project by using the alga *Selenastrum* in 264 tests. Phosphorus alone was the most limiting nutrient in 80% of the water samples tested, N was limiting in 11% of the samples and P together with N limited algal growth in 9% of the cases. In five of ten lakes examined, almost all samples showed P-limited algal growth and one lake (Esrøm) was N-limited throughout the year. The other four lakes were P-limited most of the time with a temporal shift to N- or N- and P-limitation, usually in late summer. An extreme case of temporal shift in inorganic N/ortho-P ratios was noted in Lake Tuusulanjärvi where the ratios changed from 86 in June to 0.4 and 6 in July and August.

In addition to the methodologies discussed, the inorganic N/ortho-P ratio identified at the peak of an algal bloom would be a further indicator of temporary nitrogen limitation. However, data elaboration of this aspects has not been done for the synthesis report.

4.4. CORRELATION BETWEEN BIOMASS (AVERAGE AND PEAK CHLOROPHYLL) VERSUS INLAKE PHOSPHORUS AND NITROGEN CONCENTRATIONS

For the data screening procedure applied see Appendix 3 (Procedures for Data Screening); selected statistics for the data are given in Appendix 4.

Correlation analyses for unscreened data (Table 4.1) have given correlation coefficients for average and peak chlorophyll values against total nitrogen of 0.47 and 0.50, respectively. The correlation coefficients, in both cases, are higly signigicant ($t = 3.8$, $P > 0.95$ and $t = 4.1$, $P > 0.95$). For the relationships with total phosphorus, the resulting correlation coefficients for 99 and 65 lakes respectively, were 0.75 ($t = 11.1$, $P \gg 0.99$) and 0.70 ($t = 7.8$, $P \gg 0.99$).

A re-run of the analyses on screened values (Table 4.2) considerably altered the results: the correlation co-efficients relative to nitrogen have increased to 0.64 ($t = 5.2$, $P > 0.99$) and 0.66 ($t = 5.4$, $P > 0.99$), respectively, for average and peak chlorophyll, and relative to phosphorus to 0.88 ($t = 16.1$, $P \gg 0.99$) and to 0.90 ($t = 14.3$, $P \gg 0.99$) for the corresponding variables.

Accordingly, chlorophyll is highly correlated to both inlake nitrogen and inlake phosphorus concentration, although the greater association of chlorophyll with phosphorus is evident. However, the difference of the simple correlation coefficients between phosphorus and nitrogen for the unscreened values is significant at the $P > 0.9$ level, yet for the screened values the level of significance for the difference increased to $P > 0.99$.

The procedure used to differentiate between correlation coefficients (chlorophyll versus nitrogen and chlorophyll versus phosphorus), based on t-value and u-value calculations, is not entirely valid on statistical grounds. In order to account for the high interdependency of phosphorus versus nitrogen ($r = .75$, $t = 8.4$), partial correlation coefficients have also been calculated (Tables 4.1 and 4.2).

Partial correlation coefficients result in only a slight reduction of the chlorophyll-phosphorus relationship, whereas the apparent correlation to nitrogen vanishes in all cases to levels below required significance.

Based on the pattern which evolved from these analyses, it can be concluded that, in the majority of the lakes studied in the OECD programme:

- *biomass (chlorophyll) is significantly more strongly correlated to inlake phosphorus concentrations than to nitrogen concentrations;*
- *nitrogen concentrations do not seem to play a major role in determining biomass (chlorophyll) levels.*

Table 4.1

ANALYSIS OF DEPENDENCE OF YEARLY AVERAGE AND PEAK CHLOROPHYLL, RESPECTIVELY FROM:

a) Average-Inlake Phosphorus and Nitrogen Concentrations.
b) Simple and Partial Correlation;
d) Difference between Simple Correlation Coefficients. (All cases included without Screening for Validity).
d) Correlation between Phosphorus and Nitrogen Concentrations: $r = .75$; $B\% = 56$; $t = 8.39$; $n = 57$; $P > 99$

		Average Chlorophyll		Peak Chlorophyll	
A.	$[P]_\lambda$	(Simple)	(Partial)	(Simple	(Partial)
n		99		65	
r		0.75		0.70	
B%		56%		49%	
t		11.14		7.78	
	r(ex N)		0.68		0.57
	B(ex N)%		46%		32%
B.	$[N]_\lambda$	(Simple)	(Partial)	(Simple	(Partial)
n		53		52	
r		0.47		0.50	
B%		22%		25%	
t		3.80		4.08	
	r(ex P)		-.07		-.05
	B(ex P)%		< 1%		< 1%

C. Differences Between Simple Correlation Coefficients

U	2.65	1.66
Level of Significance	$P < 0.99$	$P > 0.90$

D. Partial B% required for $P > 0.95$ for significant difference from 0 (2 variables)

N = 40	B% = 13.9
N = 60	B% = 9.5
N = 80	B% = 8.7
N = 100	B% = 6.2

These findings provide the rationale for restricting additional analyses primarily to relationships involving phosphorus (See § 5).

4.5. PEAK CHLOROPHYLL VERSUS AVERAGE CHLOROPHYLL

A highly significant correlation ($r = .95$) was found between peak chlorophyll and calculated yearly average chlorophyll (cf. Figure 6.3).

Table 4.2

ANALYSIS DIFFERENCE OF YEARLY AVERAGE AND PEAK CHLOROPHYLL RESPECTIVELY, FROM:

a) Average Inlake Phosphorus and Nitrogen Concentrations;
b) Simple and Partial Correlation;
c) Difference between Simple Correlation Coefficients (Values Selected after Screening for Validity)
d) Correlation between Phosphorus and Nitrogen as in Table 4.1

		Average Chlorophyll		Peak Chlorophyll	
A.	$[P]_\lambda$	(Simple)	(Partial)	(Simple)	(Partial)
n		77		50	
r		.88		.90	
B%		77%		81%	
t		16.05		14.30	
	r(ex N)		.79		.81
	B(ex N)%		62%		66%
B.	$[N]_\lambda$	(Simple	(Partial)	(Simple)	(Partial)
n		41		40	
r		.64		.66	
B%		41%		44%	
t		5.20		5.41	
	r(ex P)		-.06		-.04
	B(ex P)%		< 1%		< 1%
C.	Difference Between Simple Correlation Coefficients				
U		3.09		3.09	
Level of Significance		P > .99		P > .99	
D.	(as in Table 4.1)				

On average, peak chlorophyll exceeds yearly average chlorophyll by a factor of 3. However, if peak chlorophyll values are plotted against growing season averages, instead of against yearly averages, the ratio is somewhat lower (1.5 to 2) (see also Jones et al., 1979).

Analyses of the relationship in question using the extreme value theory, has further shown that the peak/average chlorophyll ratio follows Gumbel's distribution (Figure 4.5). According to this, there is 20% probability that peak values will exceed 4.5 times the annual mean, and still a 10% probability that they will exceed the annual mean by a factor of 5.5.

The findings *on peak value distribution have considerable implications for management in cases where the management objectives call for peak value rather than average chlorophyll control* (see also § 9), which may be necessary in drinking water reservoirs. In such cases, either the permissible loading objectives have to be defined accordingly, or provisions have to be made in dimensioning filter installations

Figure 4.5

CHLOROPHYLL : PEAK VALUE/AVERAGE VALUE RATIO ACCORDING TO GUMBEL'S EXTREME VALUE THEORY

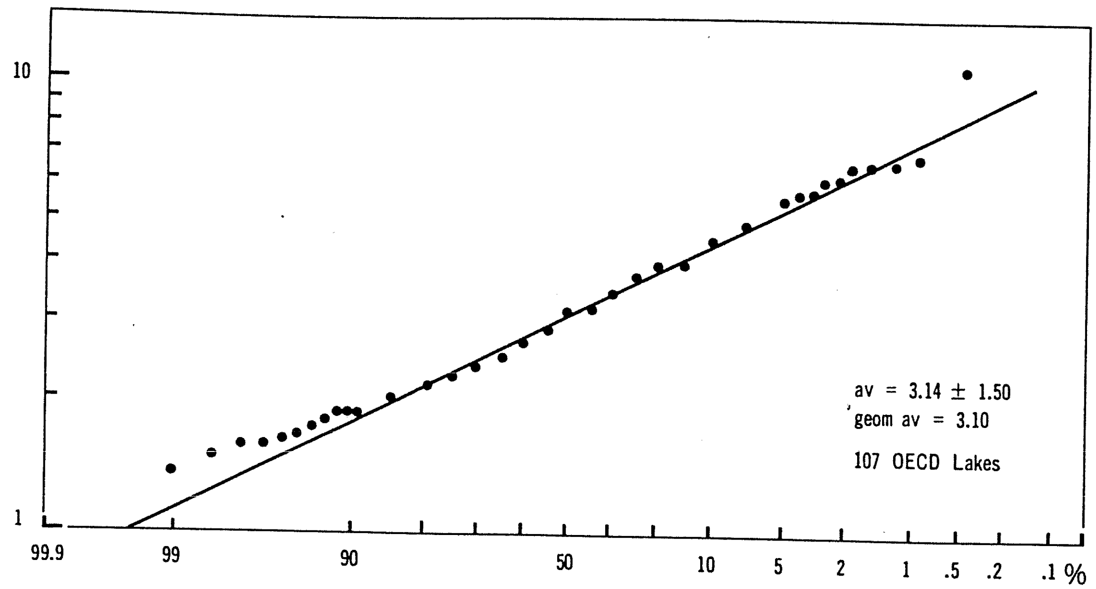
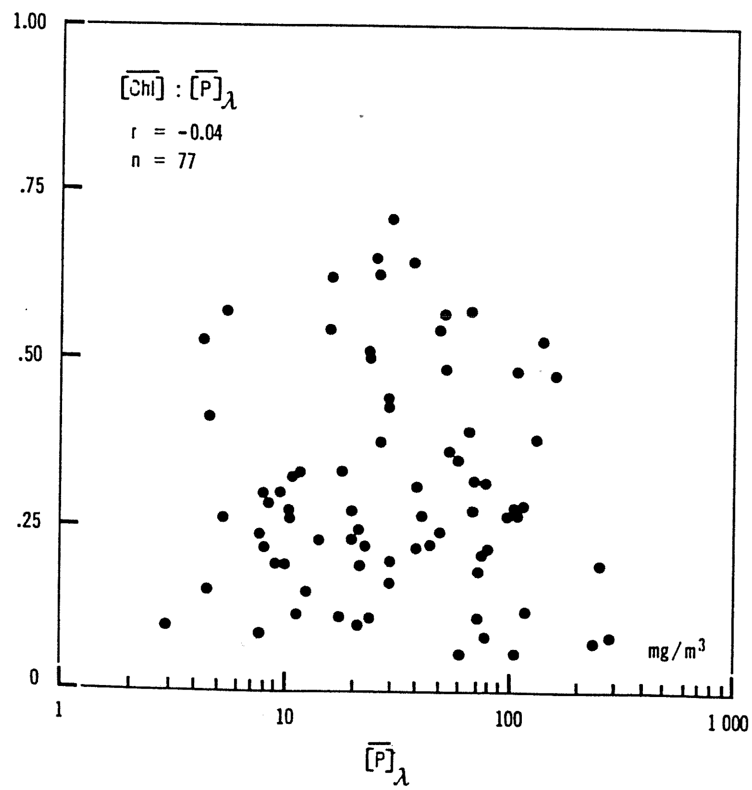


Figure 4.6



so as to cope with the 80% probability of potential peak value occurences (i.e. peak values of twice the average).

4.6. ANNUAL CHLOROPHYLL/TOTAL PHOSPHORUS RATIO VERSUS TOTAL PHOSPHORUS CONCENTRATION

This ratio varies from less than 0.1 about 0.7 for average chlorophyll, (Fig.4.6) and from 0.25 to 2.6 for peak chlorophyll (Figures 4.6 and 4.7). The plotting of a histogramme has shown that the values are of non-normal distribution. The resulting geometric averages of 0.24 for average chlorophyll and 0.65 for peak chlorophyll, agree with the result presented in § 4.5 (Figure 4.7).

No relationship exists between the ratio values in question and phosphorus concentrations. The slight tendency for peak values to increase with increasing concentration ($r = 0.11$) is not statistically significant.

The OECD finding that the chlorophyll /total phosphorus ratio does not increase with increasing inlake total phosphorus concentration, differs from that shown in earlier studies (Figure 4.8). This apparent discrepancy is probably caused by the different methodologies and data selection used in the various studies. For example, Sakamoto's (1966), which shows the greatest departure from the OECD relationship, is based on growing season and not annual mean values and it includes chlorophyll pigments other than chlorophyll *a* and does not include correction for phaeophytin.

4.7. TRANSPARENCY VERSUS CHLOROPHYLL AND VERSUS PHOSPHORUS CONCENTRATIONS, RESPECTIVELY

A significant inverse correlation has been found between yearly average Secchi disc transparency and yearly average chlorophyll ($r = -.75$; $t = 10.99$), and between Secchi disc transparency and yearly average total phosphorus concentrations ($r=-.47$; $t = 5.09$). In general, valid relationship of this sort cannot be expected. Secchi disc transparency, besides depending on biomass, in many cases is also dependent on colour and/or mineral turbidity. Several reservoirs of the OECD programme exhibit high mineral turbidity and some lakes have high water colour. The influences of mineral turbidity and colour on transparency could not be differentiated because of lack of quantitative information.

4.8. PRIMARY PRODUCTION VERSUS PHOSPHORUS AND CHLOROPHYLL CONCENTRATIONS, RESPECTIVELY

Significant correlations on a log-log linear scale, have been found between yearly planktonic primary production (expressed as g carbon/m^2.year) and annual average inlake total phosphorus concentration ($r = .71$, $n = 57$) and annual mean chlorophyll *a* concentration ($r = .74$, $n = 57$), respectively. Depending on the effect of light absorption (including self-shading effects) the relationship is non-linear (see § 6.3). Therefore, the correlations calculated do not consider the self-shading effects of algae.

Figure 4.7

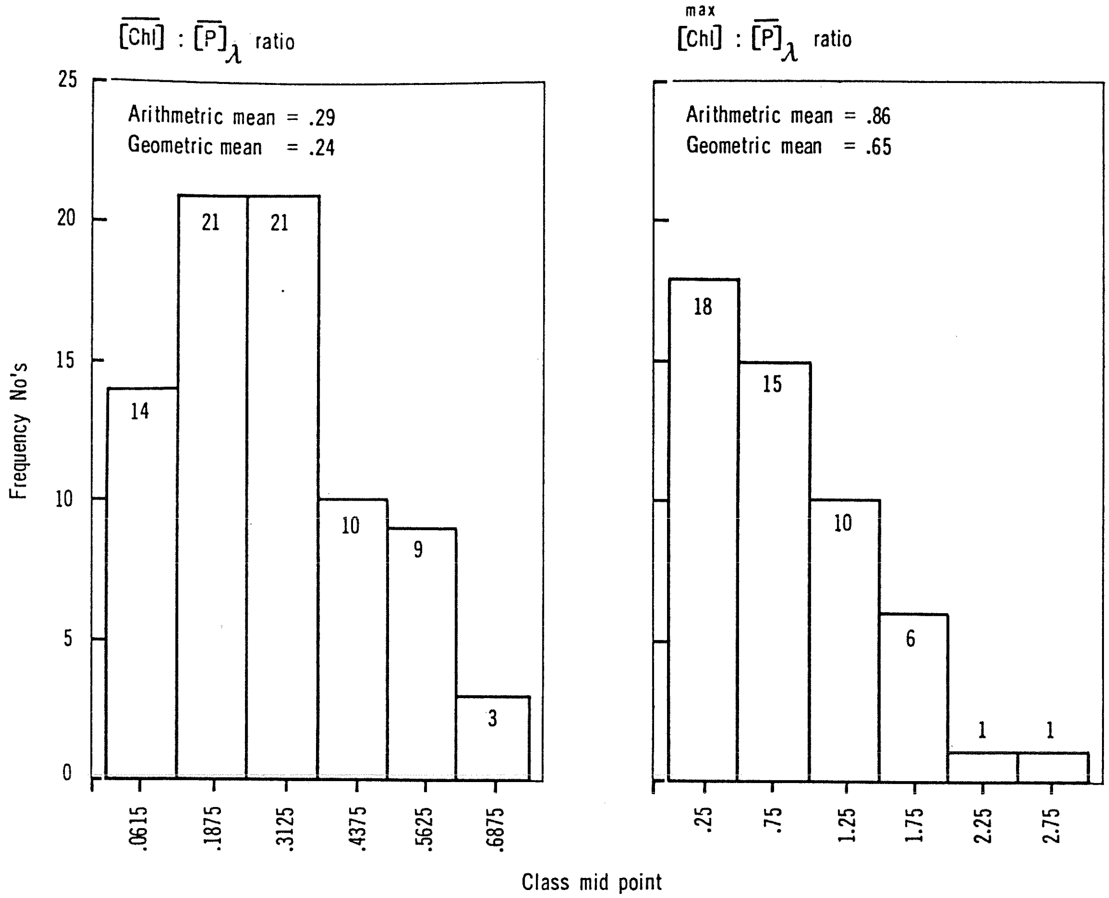
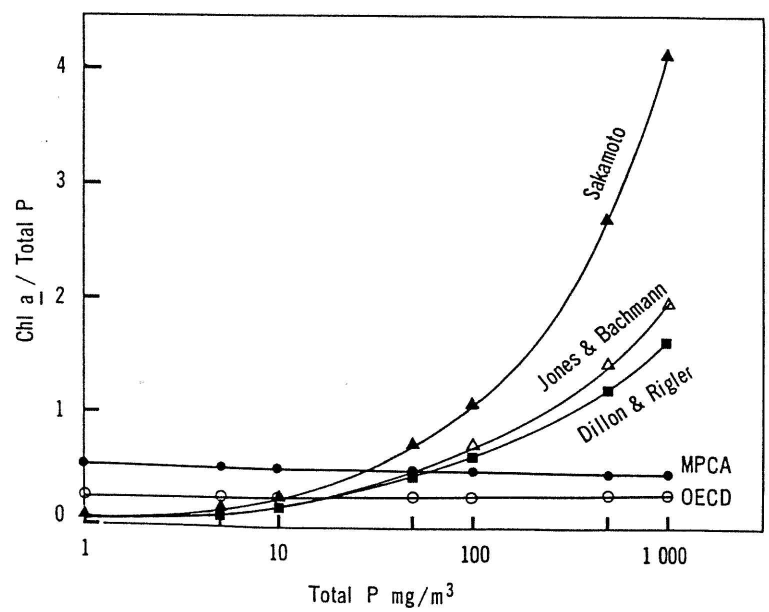


Figure 4.8



5. CORRELATIONS BETWEEN TROPHIC INDICATORS AND LOADING

5.1. THE $[M]_\lambda/[M]_j$ RATIO

This ratio (mean lake concentration/mean inflow concentration) is of considerable *a priori* diagnostic value, and has been chosen as the criterion for the selection of lakes which are to be included or excluded from correlation analyses under the above heading. This is based on the assumption that under normal conditions, the ratio in question for a non-conservative substance is $\leqslant 1$.

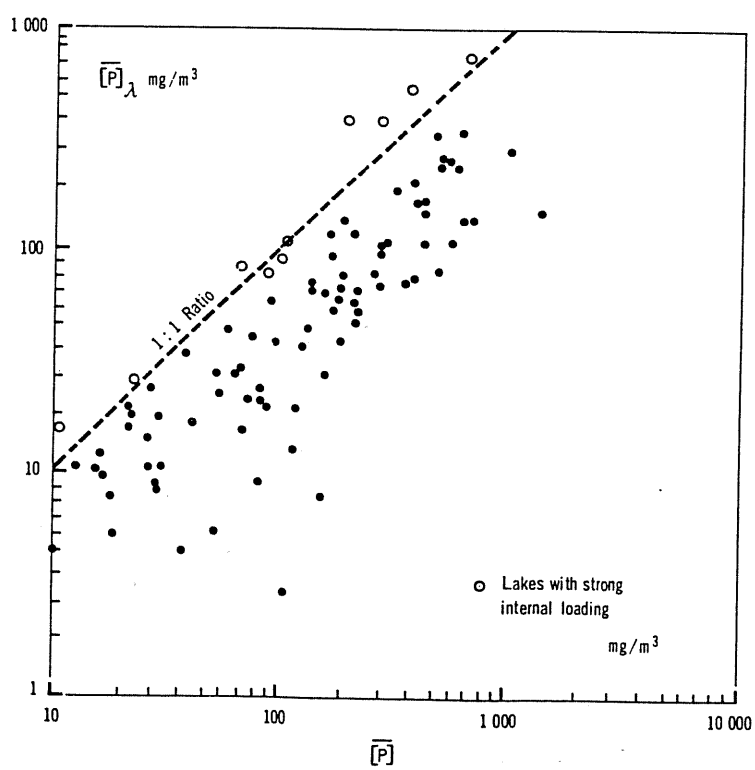
A ratio of > 1 may result for two essentially different reasons:

a) External loading has been considerably underestimated or average inlake concentration is overestimated, and hence, this information may not be usable;

b) External loading and inlake concentration have been correctly estimated but internal loading is an important nutrient source. This may be the case particularly when the lake is in a phase of washout, following external load reduction.

Data have been screened for such situations and questionable cases have been excluded from specific correlations (Figure 5.1).

Figure 5.1



5.2 STANDARD CORRECTION FOR FLUSHING

In cases where flushing has been included as a parameter, a standard function,

$$\frac{1}{1+\sqrt{\bar{z}/q_a}} = \frac{1}{1+\sqrt{T(w)}}$$

derived by Vollenweider (1976) has been used for a first approximation. Also a more general function, i.e.

$$\frac{1}{1 + a.T(w)^b}$$

has been tested. This modification produced no substantial improvement, except for in the Reservoir project. In this latter project iteration procedures applied to data, including values removed under § 3.6 et § 5.1 for the synthesis report, gave values significantly different from 1, for a, whereas b remained close to 0.5.

5.3. CORRELATION BETWEEN PHOSPHORUS AND NITROGEN LOADINGS AND CORRESPONDING INLAKE CONCENTRATIONS

Significant correlations have been found between nutrient loading and inlake nutrient concentrations. The situation varies slightly in the various projects.

A high correlation for the postulated relationship for *phosphorus* was found in all projects using only average inflow concentration as reference ($r = 0.73$ to 0.92) (Appendix 3). Inclusion of flushing or flushing and mean depth, only marginally improved the correlation ($r = 0.86$ to 0.95) in individual projects.

For the combined OECD data ($N = 87$), the correlation coefficients are .87 ($t = 16.2$) and .93 ($t = 23.3$), and both are highly significant. The difference between the two correlations ($U = 2.28$ $P > .97$) is not worthy and sufficiently significant for the flushing corrected inflow concentration to be adopted as a standard reference in further analyses.

A correspondingly high correlation has been found between flushing corrected inflow concentration and inlake concentration with regard to nitrogen ($r = .92$; $t = 14.8$), although there are only about half the available data points ($n = 42$) as for the phosphorus relationship.

5.4 CORRELATION BETWEEN TROPHIC PARAMETERS AND PHOSPHORUS AND NITROGEN LOADINGS, RESPECTIVELY

Significantly high correlations have been found between biomass parameters (average chlorophyll, peak chlorophyll) and flushing corrected phosphorus and nitrogen loadings. In addition, significant correlations between flushing corrected phosphorous loadings and metabolic parameters (primary production, hypolimnetic oxygen depletion) could be established for part of the programme.

Phosphorus. The correlation coefficients found between yearly average and peak chlorophyll versus flushing corrected phosphorus loading amounts to .78 ($t = 11.8$, $n = 91$) and .77 ($t = 9.0$, $n = 58$) respectively, for the combined values; no significant differences between the projects exist.

Nitrogen. The corresponding correlation coefficients relative to nitrogen loadings ($r = .60$; $t = 5.1$; $n = 48$; $r = .65$; $t = 5.7$; $n = 47$) are somewhat lower than for phosphorus, yet are still highly significant.

Phosphorus versus Nitrogen Dependency

In order to distinguish the dependency of chlorophyll on either factor, a corresponding analysis of the data has been made as reported in 4.4. (Tables 5.1 and 5.2). Also in this case - rather than attempting an uncontestable analysis — the discrimination is based on the common pattern evolving from various statistical treatments of the essential information.

The result of this corresponds exactly with that reported in § 4.4. Chlorophyll, (both, yearly averages and peak values), is more significantly correlated to phosphorus than to nitrogen, and, by accounting for the correlation between phosphorus and nitrogen loadings, the apparent correlation to nitrogen loading vanishes to levels of nearly zero.

Therefore, considering all that has been said previously, it can be concluded that in the majority of cases studied in the OECD programme, the production level of these lakes is controlled by phosphorus, not by nitrogen.

This is the most important confirmation obtained from the OECD study, and all further considerations in this report are based on this.

Therefore, further analyses, and use of the respective results, will be restricted largely to the phosphorous aspect.

5.5. PRIMARY PRODUCTION VERSUS PHOSPHORUS LOADING

A significant correlation on a log-log linear scale, has been found between yearly planktonic primary production (expressed as g carbon/m^2.year) and flushing corrected phosphorus loading ($r = .72$, $n = 42$). However, because of the effect of light absorption (including self-shading effects) the relationship is non-linear (see § 6.3). Therefore, the calculated correlation does not consider the self-shading effects of algae.

5.6. HYPOLIMNETIC OXYGEN DEPLETION

A comprehensive treatment of the available data is incomplete and substantial difficulties have been encountered with regard to this parameter. Preliminary analyses of the relationship between oxygen depletion rates and phosphorus loading indicate that a correlation exists at the level of about $r = 0.6$ for lakes of the US - North American Project and the Nordic Project. In the latter case P is significantly above 0.95.

For reasons which are not yet clear, the data collected in the Alpine Project remain inconclusive with regard to this relationship which is in contrast to experience from current studies. In view of the importance of hypolimnetic oxygen depletion in relation to trophic condition, this matter requires urgent attention.

In relation to reservoirs, further major difficulties arise in attempting to correlate hypolimnetic oxygen depletion rate with nutritional and/or trophic indicator parameters. This stems from the different ways by which water is withdrawn from the various reservoirs. In some cases water withdrawal comes from the epilimnion, in others from the hypolimnion, or in between. Also, in some of the reservoirs studied, hypolimnetic aeration has been installed while others are artificially mixed. Therefore, no direct comparison between reservoirs in relation to their hypolimnetic oxygen conditions is possible.

Tableau 5.1

ANALYSIS OF DEPENDENCE OF :

a) yearly average and peak chlorophyll respectively, from flushing corrected phosphorus and nitrogen loadings;
b) simple and partial correlations;
c) difference between simple correlations coefficients; (all cases included).
d) correlation between phosphorus and nitrogen loadings: $n = 48$; $r = .82$; $B\% = 67$; $t = 9.72$; $P > 99$

	Average chlorophyll		Peak chlorophyll	
A. $[\overline{P}]_j / 1 + \sqrt{T(w)}$	(Simple)	(Partial)	(Simple)	(Partial)
n	91		58	
r	.78		.77	
B%	61%		59%	
t	11.78		8.99	
r(ex N)		.63		.54
B(ex N)%		40%		29%
B. $[\overline{N}] / 1 + \sqrt{T(w)}$	(Simple)	(Partial)	(Simple)	(Partial)
n	48		47	
r	.60		.65	
B%	36%		42%	
t	5.09		5.72	
r(ex P)		−.11		−.05
B(ex P)%		< 1%		< 1%

C. Difference between simple correlation coefficients

	Average chlorophyll	Peak chlorophyll
U	1.95	1.21
Level of Significance	P > .94	P > .75

D. Partial B% required for P > .95 for significant difference from 0 (2 variables)

N = 40	B% = 13.9
N = 60	B% = 9.5
N = 80	B% = 8.7

Table 5.2

ANALYSIS OF DEPENDANCE OF:

a) yearly average and peak chlorophyll respectively from flushing corrected phosphorus and nitrogen loadings;
b) simple and partial correlations;
c) difference between simple correlation coefficients; (values selected of lakes for which both phosphorus and nitrogen loadings are available after screening for validity);
d) correlation between phosphorus and nitrogen loadings as in Table 5.1

	Average chlorophyll		Peak chlorophyll	
A. $[\overline{P}]_j / 1+\sqrt{T(w)}$	(Simple)	(Partial)	(Simple	(Partial)
n valid	40		39	
r	.86		.88	
B%	74%		77%	
t	10.39		11.27	
r(ex N)		.72		.76
B(ex N)%		52%		58%
B. $[\overline{N}]_j / 1+\sqrt{T(w)}$	(Simple)	(Partial)	(Simple)	(Partial)
n valid	40		39	
r	.68		.69	
B%	46%		48%	
t	5.72		5.80	
r(ex P)		− .09		− .12
B(ex P)%	1%	< 1%		< 1.4%
C. Difference between simple correlation coefficients				
U	2.00		2.24	
Level of significance	P > .95		P > .97	
D. (As in Table 5.1).				

6. REGRESSIONS BETWEEN NUTRITIONAL FACTORS AND TROPHIC INDICATORS

Preamble

Bearing in mind what has been said in § 4, the regression analyses presented here as based on data which has been rigorously scrutinised for outliers to make it possible to recognize the prevailing relationships between those factors which are thought to be functionnally dependent. In this process a certain subjectivity in data selection has been unavoidable, in spite of every effort to exclude only data for which there were sufficient evidence for them to be considered as non-conforming. (Appendix 3).

Main attention has been given to phosphorus as the key factor, the rationale for which is given in § 5.

For the regression equations, there remains uncertainty as to which of the two variables is to be selected as the independent, and which as the dependent variable. This relates to the fact that with each data set, statistical random errors in the data points are connected but cannot be sufficiently quantified. Therefore, the best available procedure would be to estimate the regression constants without making the distinction between x and y required in regression analyses.

One approach to this is by estimating the variance orthogonal to the regression line. This procedure has been used in the Alpine Project, but could not be applied to the combined OECD data. However, approximate orthogonal equations are added in some equations (Appendix 5.5).

Regression equations have been calculated in each project on the basis of available data. The corresponding equations differ slighlty between the various projects, but, with the exception of the Reservoir Project, no major anomalies were found. Some of the deviating regression equations in this latter project seem to stem from the high correlation found between volume and area which make the two variables interchangeable.

6.1. REGRESSIONS BETWEEN CHLOROPHYLL AND INLAKE PHOSPHORUS

Two groups of lakes were screened out after testing for validity. Both groups appeared to be outliers on the original graph with data points well below to regression line. The first group included three reservoirs in the Shallow Lakes and Reservoirs Project (Hondred en Dertig, Pretusplaat, and Mount Bold). In these reservoirs light, rather than phosphorus is limiting algal biomass. In Mount Bold Reservoir, the light limitation is "natural" because of the heavy silt load. In the other two reservoirs, artificial circulation by air injection is used to control algal growth. The Queen Elizabeth II Reservoir is also in this category, but because only orthophosphate concentrations are available for this water body, it does not appear on the chlorophyll-inlake phosphorus regression plots.

The second group of lakes that were screened out are those where nitrogen limitation could be assumed, as indicated by inorganic-N/ortho P ratios (i.e., < 10 by weight). Nineteen lakes belong to this group including the eight basin of Lugano Lake. Most lakes in this group fell below the chlorophyll-inlake phosphorus regression line, while all lakes in this group fell below the peak chlorophyll-inlake phosphorus regression line. Lakes with severe nitrogen limitation (inorganic-N/ortho-P ratios $\leqslant 5$ by weight, i.e. Esrøm, Greifensee, Léman, Lugano, Mendota) were outliers on the unscreened data plot and fell well below the regression line. The regression equation between the unscreened and the screened data are significantly different at the $P \gg 0.95$ level and the screening process reduced the standard error of estimate from . 335 to .251 (Table 6.1). The chlorophyll-inlake phosphorus regression has an obvious diagnostic value in recognising lakes where factors other than phosphorus may be limiting algal biomass.

From the highly significant correlation between these parameters (see § 4.4), the following regression equations have been calculated for screened and unscreened data (Table 6.1). The correlation equations resulting from the screened data, which have also been plotted in Figures 6.1 and 6.2, are :

$$[\overline{chl}] = .28\ [\overline{P}]_{\lambda}^{.96}$$

and

$$\begin{bmatrix} max \\ chl \end{bmatrix} = .64\ [\overline{P}]_{\lambda}^{1.05}$$

The corresponding approximative orthogonal regression equations would be:

$$[\overline{chl}] = .18\ [\overline{P}]_{\lambda}^{1.09}$$

$$\begin{bmatrix} max \\ chl \end{bmatrix} = .42\ [\overline{P}]_{\lambda}^{1.17}$$

Table 6.1.

REGRESSION EQUATIONS RELATING YEARLY AVERAGE AND PEAK CHLOROPHYLL, RESPECTIVELY, WITH AVERAGE INLAKE TOTAL PHOSPHORUS CONCENTRATION AND PEAK CONCENTRATION AND PEAK CHLOROPHYLL FROM YEARLY AVERAGE VALUE

	1 All points included without for validity	2 Values selected after screening for validity	3 Same as 2.plus values with yearly average inorganic N/ orthophosphate-P ratios < 10 excluded
A	$[\overline{chl}] = .61\ [\overline{P}]_{\lambda}^{.69}$ n = 99; r = .75; SE = .335	$[\overline{chl}] = .38\ [\overline{P}]_{\lambda}^{.86}$ *n = 88; r = .86; SE = .272	$[\overline{chl}] = .28\ [\overline{P}]_{\lambda}^{.96}$ n = 77; r = .88; SE = .251 $[\overline{chl}] = .18\ [\overline{P}]_{\lambda}^{1.09}$ (orthogonal equation)
B	$\begin{bmatrix} max \\ chl \end{bmatrix} = 1.77 [\overline{P}]_{\lambda}^{.67}$ n = 65; r = .70; SE = .375	$\begin{bmatrix} max \\ chl \end{bmatrix} = .90 [\overline{P}]_{\lambda}^{.92}$ *n = 54; r = .86; SE = .296	$\begin{bmatrix} max \\ chl \end{bmatrix} = .64 [\overline{P}]_{\lambda}^{1.05}$ n = 50; r = .90; SE = .257 $\begin{bmatrix} max \\ chl \end{bmatrix} = .42\ [\overline{P}]_{\lambda}^{1.17}$ (orthogonal equation)
C	$\begin{bmatrix} max \\ chl \end{bmatrix} = 2.86\ [\overline{chl}]^{1.03}$ n = 73; r = .93; SE = .199	$\begin{bmatrix} max \\ chl \end{bmatrix} = 2.60\ [\overline{chl}]^{1.06}$ n = 72; r = .95; SE = .167	

* r = correlation coefficient, SE = standard error of estimates, n = number of data points.
** The eight Lugano Lake basins are not included.

Figure 6.1

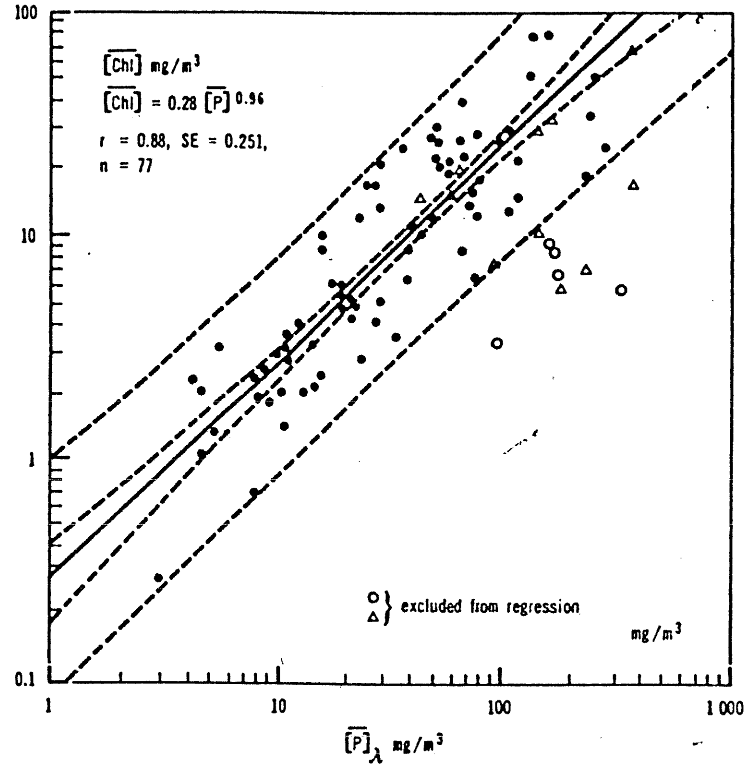


Figure 6.2

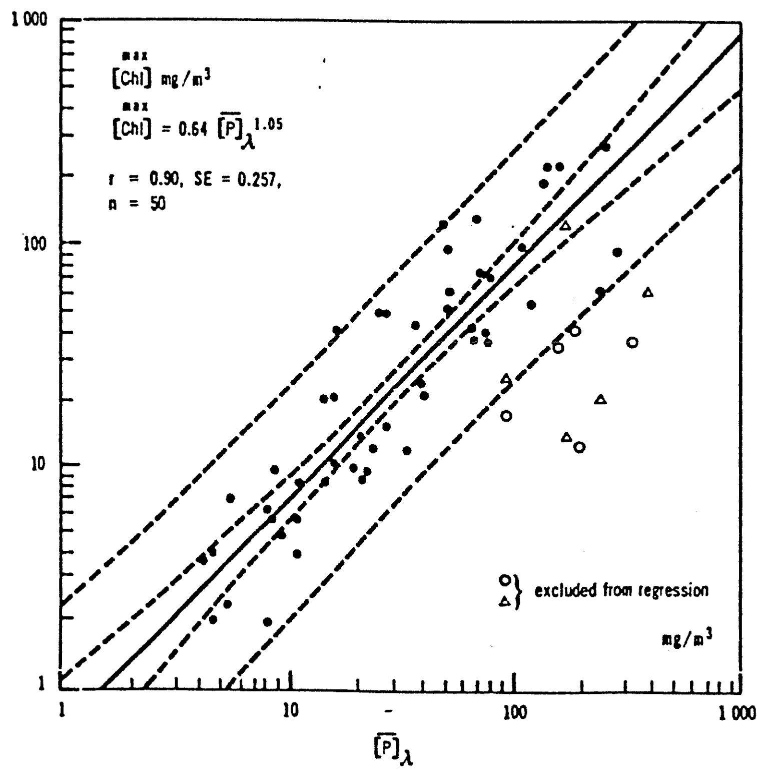


Table 6.2

REGRESSION EQUATIONS, RELATING ANNUAL MEAN CHLOROPHYLL AN ANNUAL PEAK CHLOROPHYLL *A* CONCENTRATION WITH AVERAGE INLAKE TOTAL PHOSPHORUS CONCENTRATION FOR THE COMBINED, SCREENED OECD DATA, REGIONAL PROJECTS AND REGIONAL REPORTS

(See Note in Appendix 3)*

Project	Equation	r	SE	n	Equation	r	SE	n
Combined OECD data	$[\overline{chl}] = .28[\overline{P}]_\lambda^{.96}$	.88	.251	77	$\begin{bmatrix}max\\chl\end{bmatrix} = .64[\overline{P}]_\lambda^{1.05}$	.90	.257	50
Nordic.	$[\overline{chl}] = .08[\overline{P}]_\lambda^{1.27}$	.93	.238	14	$\begin{bmatrix}max\\chl\end{bmatrix} = .25[P]_\lambda^{1.29}$	.94	.226	14
Alpine.	$[\overline{chl}] = .48[\overline{P}]_\lambda^{.87}$	.83	.279	16	$\begin{bmatrix}max\\chl\end{bmatrix} = .78[\overline{P}]_\lambda^{1.07}$	.90	.250	16
Shallow L. & R. .	$[\overline{chl}] = .52[\overline{P}]_\lambda^{.81}$	.90	.217	21	$\begin{bmatrix}max\\chl\end{bmatrix} = .74[\overline{P}]_\lambda^{.97}$	.91	.248	20
U.S.A..	$[\overline{chl}] = .20[\overline{P}]_\lambda^{1.04}$	.91	.226	26	–	–	–	–
Shallow L.& R. Report	$[\overline{chl}] = .43[\overline{P}]_{eu}^{.88}$	–	–	52	$\begin{bmatrix}max\\chl\end{bmatrix} = .57[\overline{P}]_e^{1.07}$	–	–	42

* r = correlation coefficient, SE = standard error of estimate, n = number of data points.

Regressions of log transformed data between annual mean chlorophyll in inlake phosphorus concentrations were not carried out in the regional reports. Regression equation equations for each individual project were calculated from the combined data using the screened data (Table 6.2). The slopes of the regression for $[\overline{chl}] - [\overline{P}]_\lambda$ and $\begin{bmatrix}max\\chl\end{bmatrix} - [\overline{P}]_\lambda$ for the individual projects range from .81 (SLR) to 1.27 (Nordic) and .96 for the combined OECD data, and from .97 (SLR) to 1.29 (Nordic) and 1.05, respectively. The slopes of the regressions are not significantly different at the $P > 0.95$ level. The higher slope for the Nordic Project is probably due to the different methodology used in this project, in which chlorophyll *a* values were not corrected for phaeophytin: if such corrections had been made in this project, the slope of the regression would have been considerably lower and it would have been slightly lower for the combined OECD data.

6.1.1. Regression between Peak and Annual Mean Chlorophyll

The regression equation between peak chlorophyll and annual mean chlorophyll values have been calculated (Figure 6.3, Table 6.1):

$$\begin{bmatrix}max\\chl\end{bmatrix} = 2.60\ [\overline{chl}]^{1.06}$$

Peak chlorophyll exceeds yearly average chlorophyll by a factor of 2.6 with an increasing trend for higher peak values at higher annual mean concentrations which is consistent with § 4.5.

The peak chlorophyll-annual mean chlorophyll relationships presented here should be considered with caution. The annual mean chlorophyll values were obtained from sampling of different frequency in various lakes, thus the annual mean value reflects more or less the true mean value for individual lakes. In contrast, the true peak value would often be underestimated in lakes which were sampled less frequently.

Figure 6.3

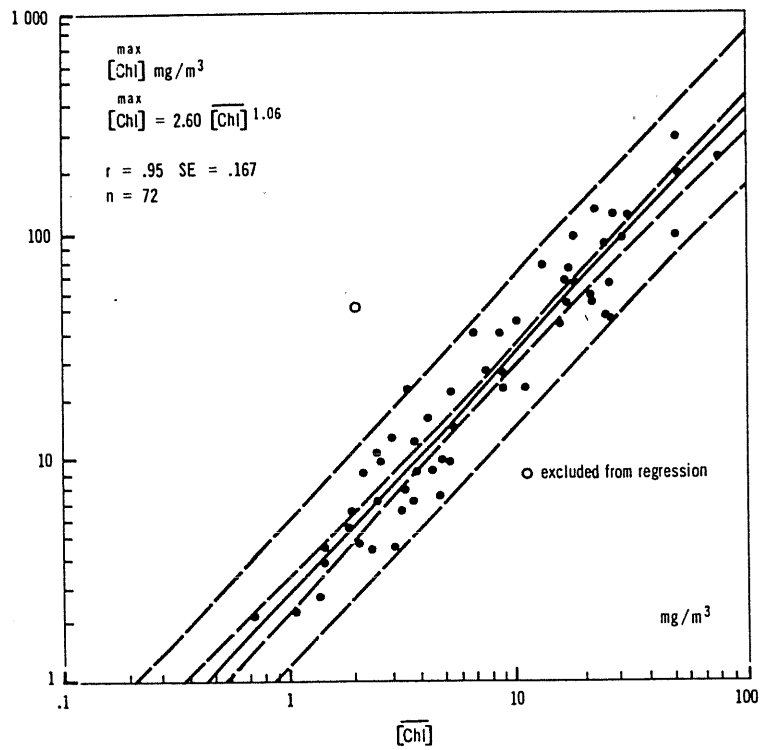
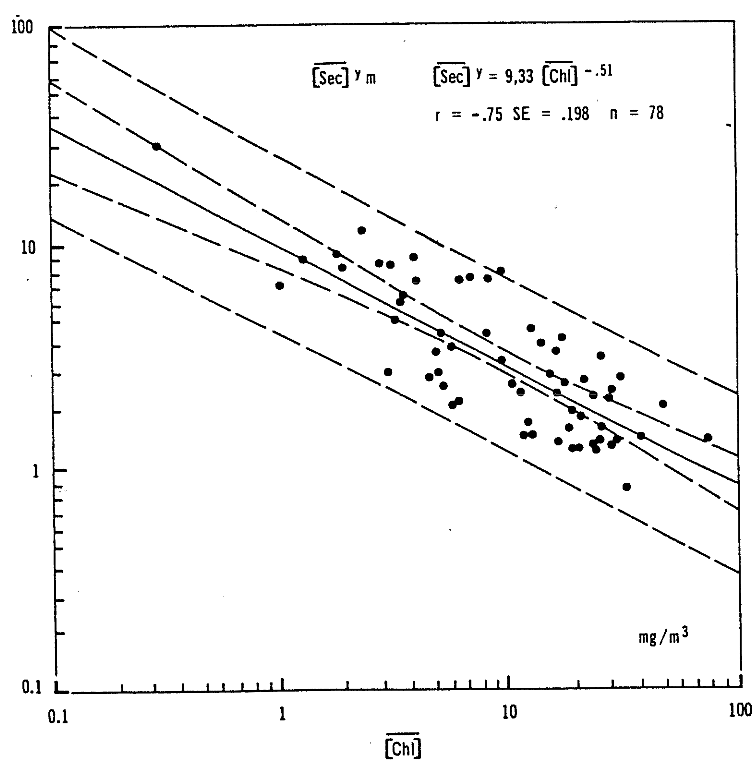


Figure 6.4



Queen Elizabeth II Reservoir (Shallow Lakes and Reservoirs Project) with a peak chlorophyll value of 47.5 mg/m^3 and annual mean value of 2.0 mg/m^3 is a special case (Figure 6.3). The reason for this is that algal biomass is suppressed by artificial circulation (see § 6.1). Based on the annual mean orthophosphate concentration of 892 mg PO_4-P/m^3 which is the highest in the OECD study, without intervention the peak and annual mean chlorophyll values would have been considerably higher.

6.2. SECCHI TRANSPARENCY AGAINST CHLOROPHYLL AND INLAKE PHOSPHORUS

The Secchi disc transparency, depends on both biotic extinction wich is proportional to chlorophyll concentration, and on abiotic light extinction caused by mineral turbidity (stilt) and water colour (dissolved humic substances). For this report, it was not possible to screen the data for abiotic influences. Consequently, the relationships including Secchi disc transparency, should be viewed with caution; they could be improved with further work and be careful screening of the data.

Although yearly Secchi disc transparency is significantly correlated to both yearly average chlorophyll and total phosphorus concentrations (see § 4.7), only the first relationship can be functionally related.

6.2.1. Secchi Transparency versus Chlorophyll

Based on all available data, average annual Secchi disc transparency depends on chlorophyll according to (Figure 6.4)

$$\overline{[Sec]}^{y} = 9.33\ \overline{[chl]}^{-.51} \qquad r = .75,\ SE = .198,\ n = 78$$

By using the converse of this relationship, a rough estimate of the average chlorophyll level would be possible in lakes in which substantial amounts of mineral turbidity and/or high colour are absent:

$$\overline{[chl]} = 30.8\ \overline{[Sec]}^{-1.11}$$

This is important because for many lakes, historical data is available on Secchi disc transparency but not on chlorophyll. Applied with caution, then the historical evolution, in terms of trophic changes, can be approximated.

In the USA project the Secchi disc transparency-chlorophyll relationship was explored including data from the literature, some of which were not corrected for phaeophytin; by combining ice-free season, summer, and annual values, the following relationship was obtained:

$$\overline{[Sec]} = 6.35\ \overline{[chl]}^{-.473} \qquad r = -.85$$

Because of the differences in data selection in the USA project and in this report, the two relationships are not directly comparable. The same comment would apply to the summer Secchi disc transparency-summer chlorophyll relationship (not corrected for phaeophytin) which gave the following equation in the Nordic report:

$$\overline{[Sec]}^{Su} = 4.74\ \overline{[chl]}^{Su\text{-}.267} \qquad r = .65$$

6.2.2. Secchi Transparency versus Total Phosphorus

Secchi disc transparency is also related to phosphorus, for which the following equation was obtained (Figure 6.5):

$$\overline{[Sec]}^{y} = 9.77\ \overline{[P]}_{\lambda}^{-.28} \qquad r = -.47;\ SE = .255;\ n = 87$$

Figure 6.5

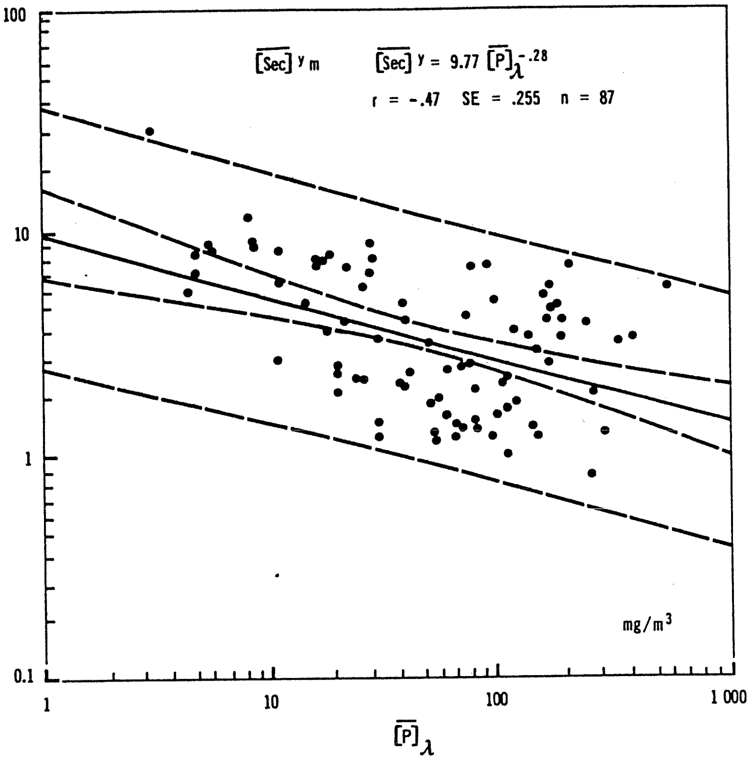
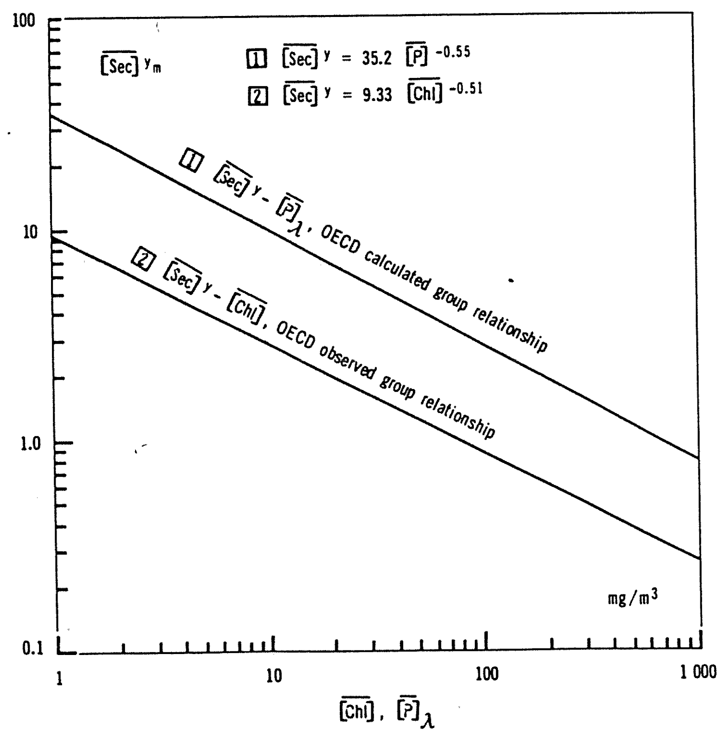


Figure 6.6



However, as Secchi disc transparency depends on phosphorus by virtue of biomass, the regressions relating these two factors, though statistically significant, have only limited meaning. To overcome this difficulty the converse of this relationship (total phosphorus versus Secchi disc transparency) may be calculated and by averaging the two equations thus obtained, the following relationship has been found (Figure 6.6):

$$\overline{[Sec]}^{y} = 35.2\ \overline{[P]}_{\lambda}^{-.55}$$

which gives a more realistic expression of the Secchi disc-total phosphorus relationship. An almost identical relationship is derived by substituting chlorophyll in the Secchi disc-chlorophyll relationship by the appropriate conversion to phosphorus using the relationship found in § 6.1 (Figure 6.1).

$$\overline{[Sec]}^{y} = 35.0\ \overline{[P]}_{\lambda}^{-.51}$$

The calculated Secchi disc-total phosphorus relationship (Figure 6.6) can be used as a guide to identify cases where the chlorophyll-phosphorus relationship departs from the average group behaviour.

6.3. YEARLY PRIMARY PRODUCTION VERSUS PHOSPHORUS AND CHLOROPHYLL CONCENTRATIONS, RESPECTIVELY

Primary production is linked to phosphorus by a hyperbolic function of the form:

$$\Sigma PP = K_1 \frac{X}{\epsilon + K_2 X},$$

in which K_1 and K_2 are constants and ϵ represents the water absorption coefficient. K_1/K_2 is an upper value (saturation plateau), and K_2 refers to the self-shading effect of phytoplankton biomass. X can be either chlorophyll, inlake phosphorus concentration, or flushing corrected phosphorus loading.

The primary production data were screened for obvious outliers including very high values (> 800 g C/m².y), which exceed the highest reliable yearly primary production values reported in the literature. After screening some uncertainty still remains about the validity of some remaining data points. This uncertainty stems from the difficulties associated with adequate measurement of annual primary production (see § 3.6.).

Two options have been tested, the linear and the theoretically more appropriate hyperbolic equations, relating yearly planktonic primary production to average inlake phosphorus and annual mean chlorophyll concentrations which lead to (Figures 6.7 to 6.10):

$\Sigma PP\ (g\ C/m^2.y) = 31.1\ \overline{[P]}_{\lambda}^{.54}$ $\quad r = .71;\ SE = .265;\ n = 49$

$\Sigma PP\ (g\ C/m^2.y) = 56.6\ \overline{[Chl]}^{.61}$ $\quad r = .79;\ SE = .242;\ n = 49$

and

$$\Sigma PP = 512\ \frac{\overline{[P]}_{\lambda}}{28.1 + \overline{[P]}_{\lambda}} \qquad r = .70;\ n = 49$$

$$\Sigma PP = 631\ \frac{\overline{[chl]}}{11.8 + \overline{[chl]}} \qquad r = .74;\ n = 49$$

The saturation plateaus of 512 and 631 g C/m².y of the hyperbolic relationships estimated from total phosphorus and chlorophyll, respectively, are not significantly different at the $P > .95$ level.

In the Alpine Project the two options tested lead to

$\Sigma PP\ (g\ C/m^2.y) = 31.2\ \overline{[P]}_{e}^{.56}$ $\quad r = .75;\ SE = .241$

(orthogonal relationship),

Figure 6.7

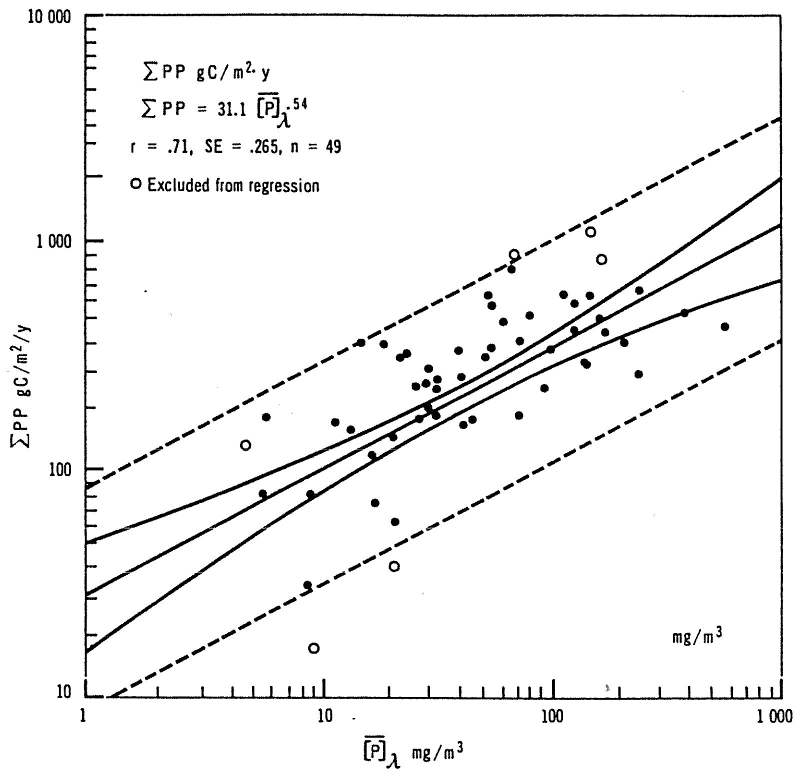


Figure 6.8

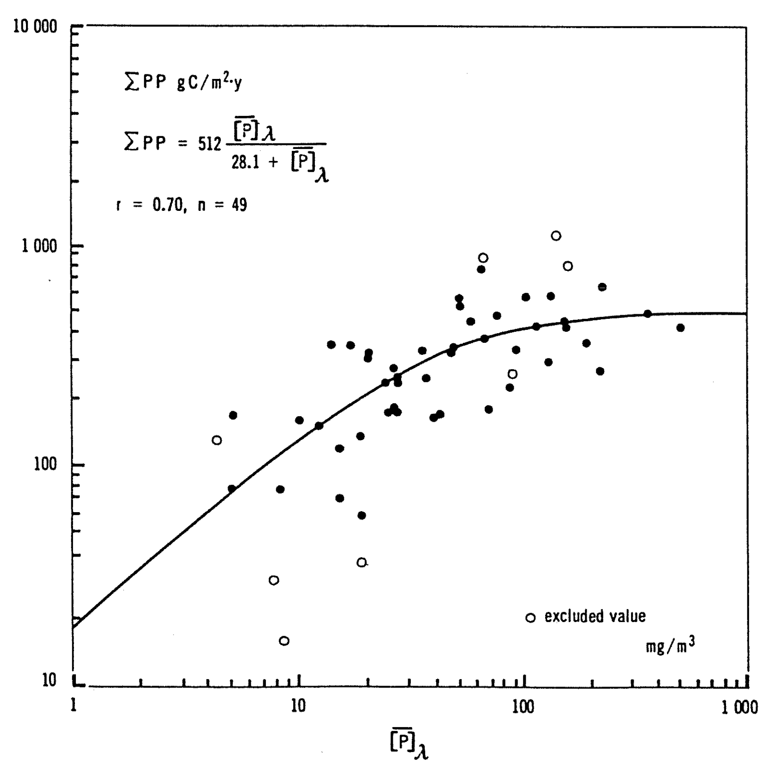


Figure 6.9

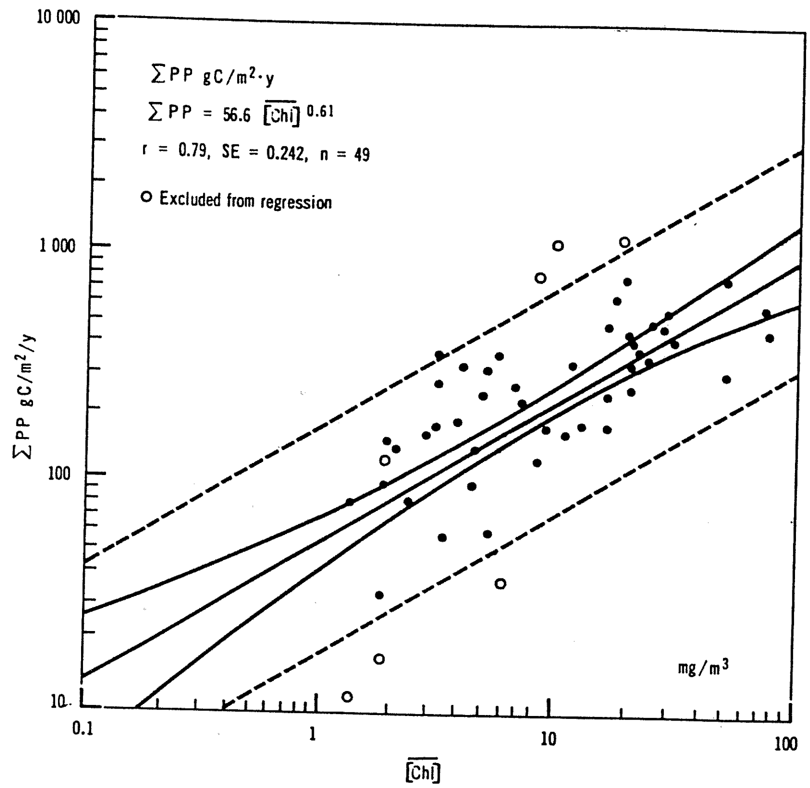
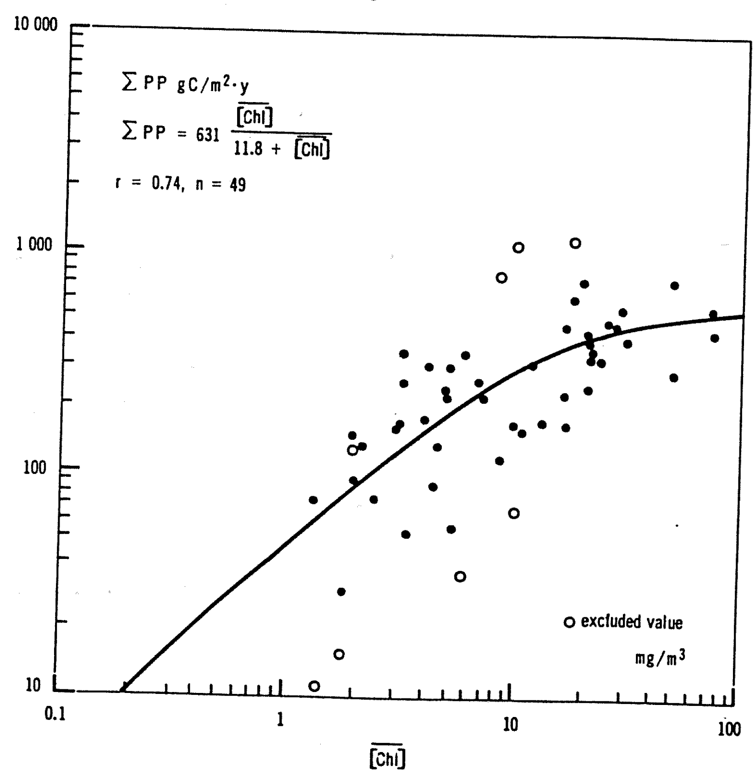


Figure 6.10



and $$\Sigma PP\ (g\ C/m^2.y) = 456\ \frac{[\overline{P}]_\lambda^{sp}}{40 + [\overline{P}]_\lambda^{sp}}$$

The saturation plateau of 456 and 512 g C/m².y of the hyperbolic relationship of the Alpine Project and of the combined OECD data respectively, are not significantly different at the $P > .95$ level.

6.4. REGRESSIONS BETWEEN TROPHIC INDICATORS AND NUTRIENT LOADINGS

As a common feature for representing this relationship, all loading terms have been expressed as yearly average inflow concentrations corrected for flushing with the standard term discussed in § 5.2.

Further modifications of this standard term have been attempted in the Reservoir Project because of the particular conditions of the water bodies studied in this project. However, in considering the overall scope of the OECD programme, no particular need was felt at this time to develop this matter further, since it was not within the scope of the programme to test models but instead to provide a basis from which management decisions can be made.

6.4.1. Total Phosphorus and Total Nitrogen Concentrations from Loadings

For phosphorus, it was found that the lake concentration can be obtained from

$$[\overline{P}]_\lambda = .76\ [\overline{P}]_j^{.83}$$

and from

$$[\overline{P}]_\lambda = .42\ [\overline{P}]_j^{.95}$$

the latter being the orthogonal relationship (Table 6.3). However, these estimates are less reliable than those in which flushing is taken into account (cf. § 5.3). The regression equations obtained for the regional projects are not significantly different at the $P > .95$ level.

Yearly average phosphorus and nitrogen concentrations have been found to be related to flushing corrected inflow concentrations as follows, considering $[M]_j/1+\sqrt{T(w)'}$ as independent variable (Figures 6.11 and 6.12, Table 6.4).

$$[\overline{P}]_\lambda = 1.55\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.82}$$

and

$$[\overline{N}]_\lambda = 5.34\left[[\overline{N}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.78}$$

The omission of lakes with internal loading (see § 5.1) and reservoirs where the inlake concentration is not representative (see § 3.6) greatly reduces the estimated standard errors (Table 6.4).

The corresponding approximative orthogonal regression equations would be

$$[\overline{P}]_\lambda = 1.22\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.87}$$

$$[\overline{N}]_\lambda = 3.25\left[[\overline{N}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.85}$$

In both regression equations, the exponent deviates significantly from 1.0 ($P \gg .99$). Therefore, the standard flushing corrected inflow concentration term as such, i.e. without inclusion of the constants derived its calculation, should not be used as a predictor for inlake concentrations. This result of the OECD study should be considered as an essential modification compared with the way in which the standard flushing correction term has been used in former studies (Vollenweider 1976, 1979; Rast and Lee 1978).

Figure 6.11

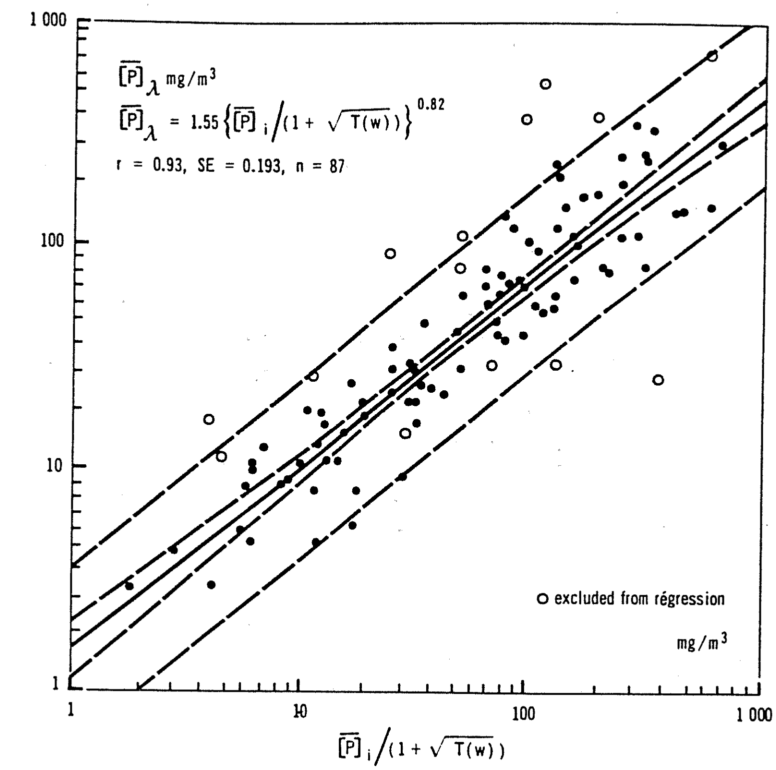


Figure 6.12

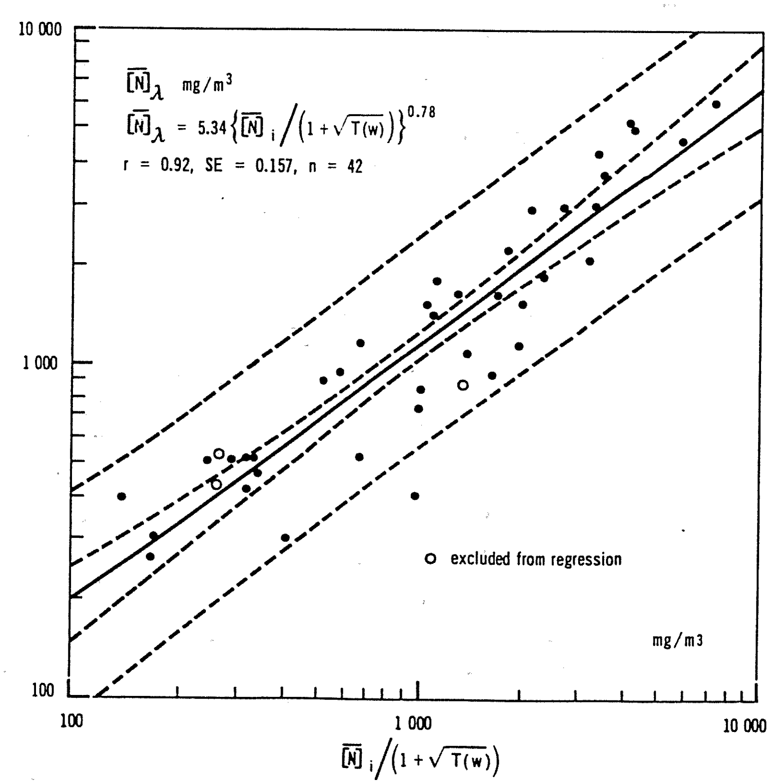


Table 6.3

REGRESSION EQUATIONS, RELATING ANNUAL MEAN INLAKE CONCENTRATIONS OF PHOSPHORUS WITH ANNUAL MEAN INFLOW CONCENTRATIONS FOR THE COMBINED, SCREENED OECD DATA, REGIONAL PROJECTS AND REGIONAL REPORTS
(see Note in Appendix 3)*

Project	Equation	r	SE	n
Combined OECD data	$\overline{[P]}_\lambda = .76\,\overline{[P]}_j^{.83}$	.87	.261	87
Nordic.	$\overline{[P]}_\lambda = 1.26\,\overline{[P]}_j^{.70}$	.73	.337	14
Alpine.	$\overline{[P]}_\lambda = .86\,\overline{[P]}_j^{.80}$	.92	.223	18
Shallow Lakes and Reservoirs	$\overline{[P]}_\lambda = .65\,\overline{[P]}_j^{.88}$	.91	.231	24
U.S.A..	$\overline{[P]}_\lambda = .66\,\overline{[P]}_j^{.85}$	.84	.286	31
S.L. and R. Report	$\overline{[P]}_\lambda = .77\,\overline{[P]}_j^{.85}$	.89	.253	41
Alpine Report.	$^{**}\overline{[P]}_\lambda = .63\,\overline{[P]}_j^{.87}$	.85	.287	–

* r = correlation coefficient, SE = standard error of estimate, n = number of data points.
** orthogonal relationship.

Table 6.4

REGRESSION EQUATIONS RELATING INLAKE YEARLY AVERAGE TOTAL PHOSPHORUS AND TOTAL NITROGEN CONCENTRATIONS WITH FLUSHING CORRECTED AVERAGE INFLOW CONCENTRATION, $\overline{[M]}_j / \left(1 + \sqrt{T(w)'}\right)$.*

	1 All points included without screening for validity	2 Values selected afted screening for validity
A	$\overline{[P]}_\lambda = .93\left[\overline{[P]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.80}$ n = 101; r = .82; SE = .314	$\overline{[P]}_\lambda = 1.55\left[\overline{[P]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.82}$ n = 87; r = .93; SE = .193 $\overline{[P]}_\lambda = 1.22\left[\overline{[P]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.87}$ (orthogonal equation)
B	$\overline{[N]}_\lambda = 5.75\left[\overline{[N]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.77}$ n = 46; r = .92; SE = .154	$\overline{[N]}_\lambda = 5.34\left[\overline{[N]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.78}$ n = 42; r = .92; SE = .157 $\overline{[N]}_\lambda = 3.25\left[\overline{[N]}_j/\left(1+\sqrt{T_{(w)'}}\right)\right]^{.85}$ (orthogonal equation)

* r = correlation coefficient; SE = standard error of estimate; n = number of data points.

The findings from the Reservoir Project are slightly at variance with the above results. It was found that the lake phosphorus concentrations are somewhat better predicted from either,

$$\overline{[P]}_\lambda = \overline{[P]}_j / \left(1 + 7\sqrt{T(w)'/\bar{z}}\right)$$

or

$$\overline{[P]}_\lambda = \overline{[P]}_j / \left(1 + 2\sqrt{T(w)'}\right)$$

Table 6.5

REGRESSION EQUATIONS, RELATING ANNUAL MEAN INLAKE PHOSPHORUS CONCENTRATIONS WITH FLUSHING CORRECTED INFLOW CONCENTRATIONS, $\overline{[P]}_j/(1+\sqrt{T(w)'}) = X$, FOR THE COMBINED, SCREENED OECD DATA, REGIONAL PROJECTS AND REGIONAL REPORTS
(See Note in Appendix 3)*

Project	Equation	r	SE	n
Combined OECD data	$\overline{[P]}_\lambda = 1.55\ X^{.82}$	.93	.192	87
Nordic.	$\overline{[P]}_\lambda = 1.12\ X^{.92}$	.86	.252	14
Alpine.	$\overline{[P]}_\lambda = 1.58\ X^{.83}$	.93	.212	18
Shallow Lakes and Reservoirs	$\overline{[P]}_\lambda = 1.02\ X^{.88}$	.95	.185	24
U.S.A..	$\overline{[P]}_\lambda = 1.95\ X^{.79}$	.95	1.60	31
Nordic Report	$\overline{[P]}_\lambda = 0.96\ X^{.96}$	.84	–	38
Alpine Report.	$^{**}\overline{[P]}_\lambda = 1.34\ X^{.88}$	.88	.232	35
Shallow Lakes & Reservoirs Report.	–	–	.333	43
U.S.A. Report.	–	–	–	–

* r = correlation coefficient, SE = standard error of estimate, n = number of data points;
** orthogonal relationship.

However, recalculation for the Shallow Lakes and Reservoirs Project using screened data gives a good prediction of inlake phosphorus concentration from the flushing corrected inflow concentration with an estimate of standard error of .185 compared with a standard error of .333 obtained in the Shallow Lakes and Reservoirs Report (Table 6.5). This discrepancy can be explained by the different methods of data treatment in the two reports. In the Shallow Lakes and Reservoirs Report, the regressions were calculated using all available data points, which included lakes with internal loading, and long, narrow reservoirs where the inlake concentration is not representative. Also included were yearly data on highly artificial pumped storage reservoirs where nutrient loading and water residence time varies greatly in different years and the total water discharge might be more or less than the annual inflow. By averaging several years of data, the method used in the calculations for this report approximates more closely a steady state condition for the pumped storage reservoirs. Contrary to what has been said in the Shallow Lakes and Reservoirs report, these bodies of water do not appear as outliers on the $\overline{[P]}_\lambda = \overline{[P]}_j/(1+\sqrt{T(w)'})$ plot. The modified equations obtained in the latter,which include bodies of water which cannot be considered to be at a steady state, only slightly reduce the estimate of the standard error to .265 and .266, respectively.

6.4.1.1. Sources of error in estimating inlake concentration of nutrients

Although the correlations between inlake concentration and flushing corrected inflow concentrations are high and significant, there remains a scatter along the regression line after data screening. This scatter is caused by inadequate estimation of the average inlake concentration with respect to outflow concentration and by the influence of sediments, in addition to methodological and sampling errors (see § 3.6).

A. *Inlake concentration*

The standard function used to estimate the annual average inlake concentration from flushing corrected inflow concentration assumes that the lake is a completely mixed reactor of constant volume with average lake concentration equal to the outflow concentration (Appendix 2).

This ideal, hypothetical condition is seldom approached in bodies of water. In lakes, the outflow concentration corresponds to that of surface water (epilimnetic), while in reservoirs water may be withdrawn at various depths (see § 5.6). However, a variety of conditions and their combination, i.e. stratification, horizontal heterogeneities, hydrologic and morphometric features, prevent complete mixing, which often results in the average inlake concentration being different from the outflow concentration.

Yearly average inlake and epilimnetic phosphorus concentrations have been found to be related to the yearly average outflow concentration as following (Figures 6.13 and 6.14, Appendix 6).

$$\overline{[P]}_o = 1.36\ \overline{[P]}_\lambda^{.92}$$

$$\overline{[P]}_o = 1.18\ \overline{[P]}_e^{.98}$$

The corresponding approximative orthogonal regression equations would be

$$\overline{[P]}_o = 1.12\ \overline{[P]}_\lambda^{.97}$$

and

$$\overline{[P]}_o = .94\ \overline{[P]}_e^{1.04}$$

The standard errors of the regression relating outflow concentration with inlake and epilimnetic concentrations of .173 et .153 respectively are somewhat lower than that of the relationship between inlake concentration and flushing corrected average inflow concentration (SE = .193), (Table 6.4). The average annual euphotic phosphorus concentration gives a more accurate estimate of the annual mean outflow concentration than the average yearly lake concentration. The errors associated with the estimation of inlake concentration, with respect to outflow concentration, undoubtly contribute to the data scatter found in § 6.4.1.

b. *Sediments*

The influence of the sediments is twofold:

a) release of phosphorus from the sediments as discussed in § 5.1;
b) direct influence of the phosphorus in the sediments.

Several models have been proposed to describe this amount, either by assuming that there is a constant fraction of the input from the sediments or a constant factor times concentration (Golterman, 1980). If this were true, no scatter should be added by the sediments. However, it seems likely that the sediments may have a more direct influence, e.g., by assuming an adsorption model. In this case, two lakes receiving the same phosphorus load, but with different *total* sediment load, will have different phosphorus concentrations.

Another possible cause of scatter is the amount of phosphorus co-precipitated with calcium carbonate ($CaCO_3$), in eutrophic hard water lakes. Thus, a hard water lake may have a lower phosphorus concentration than a soft water lake receiving the same loading.

Figure 6.13

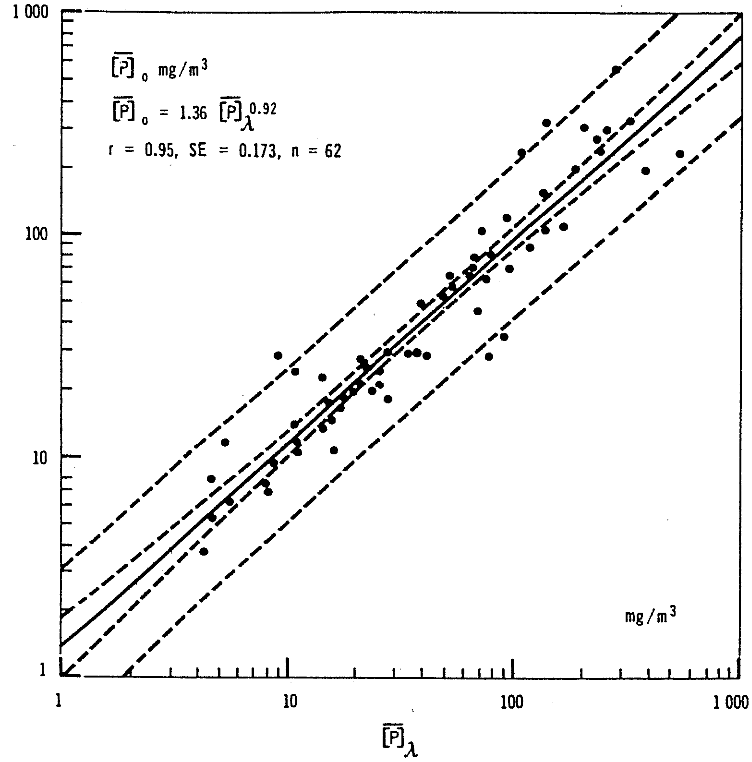
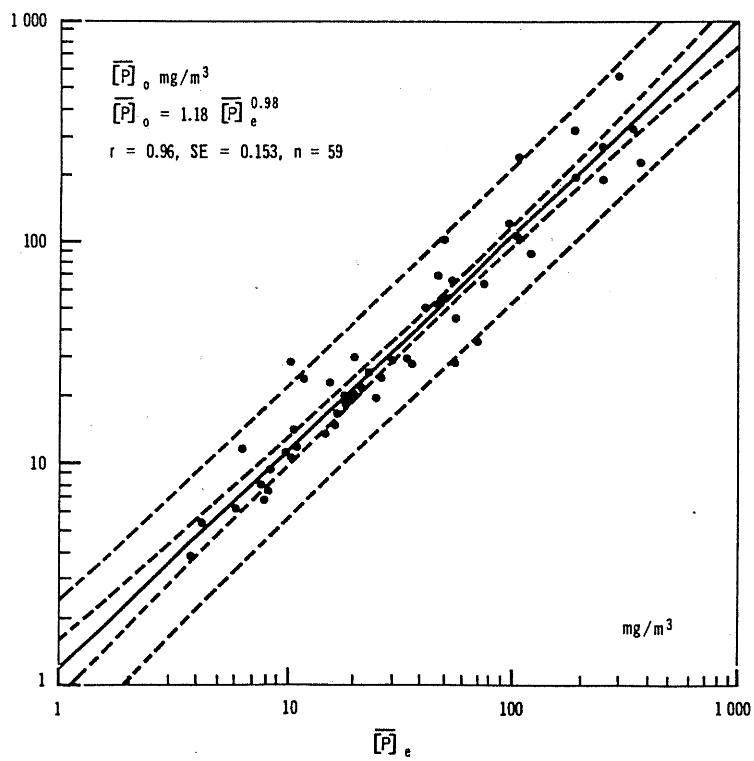


Figure 6.14



6.4.2. Regressions regarding Chlorophyll versus Phosphorus and Nitrogen Loadings, respectively

Yearly average and peak chlorophyll relate to the corresponding loading terms as follow (Figures 6.15 and 6.16, Table 6.6), whereby

$$x = \overline{[P]}_j/(1+\sqrt{T(w)'})$$

$$\overline{[Chl]} = .37\ X^{.79}$$

$$\begin{matrix}\max\\ chl\end{matrix} = .74\ X^{.89}$$

The corresponding approximative orthogonal equation would read:

$$\overline{[Chl]} \approx 0.27\ X^{.91}$$

$$\begin{bmatrix}\max\\ chl\end{bmatrix} \approx .45\ X^{1.02}$$

The deletion of points, after screening of data, as seen from Table 6.6 increases the slope of the regression and reduces the estimated standard error similar to that found in § 6.1.

The regression equations for the regional projects are given in Table 6.7. The regression equations are not significantly different at the $P > .95$ level.

With regard to nitrogen, the following equations have been calculated:

$$\overline{[chl]} = .046\left[\overline{[N]}_j/(1+\sqrt{T(w).})\right]^{.77}$$

and

$$\begin{bmatrix}\max\\ chl\end{bmatrix} = .073\left[\overline{[N]}_j/(1+\sqrt{T(w)'})\right]^{.87}$$

As follows from the discussion in § 5.4, these relations, though statistically significant, do not necessarily reflect a causal relationship in the sense of a quantitative dependency of chlorophyll from nitrogen loadings found in the lakes of the OECD

Table 6.6

REGRESSION EQUATIONS RELATING YEARLY AVERAGE AND PEAK CHLOROPHYLL CONCENTRATIONS WITH FLUSHING CORRECTED AVERAGE PHOSPHORUS INFLOW CONCENTRATION, $\overline{[P]}_j/(1+\sqrt{T_{(w)'}}) = X$.*)

	1 All points included without screening for validity	2 Values selected after screening for validity	3 Same as 2. plus values with yearly average inorganic N/orthophosphate ratios < 10 excluded
A	$\overline{[chl]} = .66\ X^{.64}$ $n = 91;\ r = .78;\ SE = .323$	$\overline{[chl]} = .40\ X^{.77}$ $n = 74;\ r = .87;\ SE = .258$	$\overline{[chl]} = .37\ X^{.79}$ $n = 67;\ r = .88;\ SE = .257$ $\overline{[chl]} = .24\ X^{.91}$ (orthogonal equation)
B	$\begin{bmatrix}\max\\ chl\end{bmatrix} = 1.73\ X^{.66}$ $n = 58;\ r = .77;\ SE = .348$	$\begin{bmatrix}\max\\ chl\end{bmatrix} = .75\ X^{.89}$ $n = 47;\ r = .88;\ SE = .287$	$\begin{bmatrix}\max\\ chl\end{bmatrix} = .74\ X^{.89}$ $n = 45;\ r = 89;\ SE = .284$ $\begin{bmatrix}\max\\ chl\end{bmatrix} = .45\ X^{1.02}$ (orthogonal equation)

* r = correlation coefficient, SE = standard error of estimate, n = number of data points.

Figure 6.15

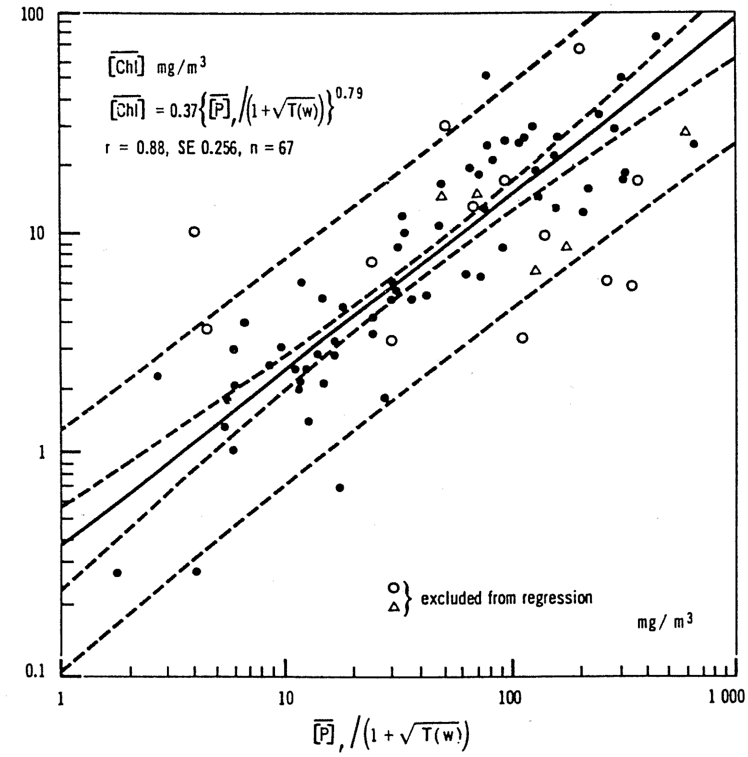


Figure 6.16

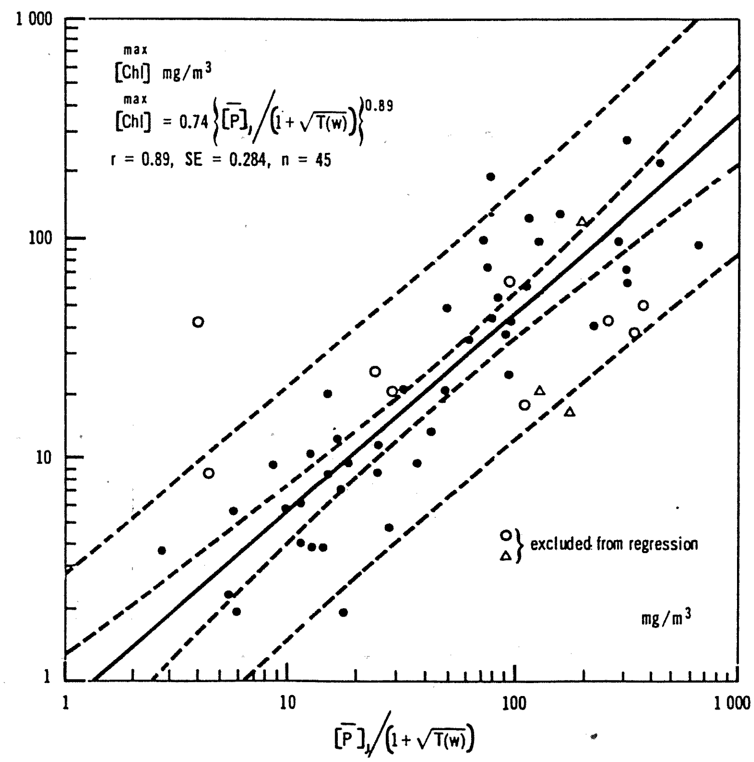


Table 6.7

REGRESSION EQUATIONS, RELATING ANNUAL MEAN AND ANNUAL MAXIMUM CHLOROPHYLL *a* CONCENTRATIONS WITH THE FLUSHING CORRECTED AVERAGE ANNUAL PHOSPHORUS INFLOW CONCENTRATIONS $[\mathrm{P}]_j/(1+\sqrt{\mathrm{T(w)}}) = \mathrm{X}$ FOR THE COMBINED, SCREENED OECD DATA, REGIONAL PROJECTS AND REGIONAL REPORTS
(see Note in Appendix 3)*

Project	Equation	r	SE	n	Equation	r	SE	n
Combined OECD data	$\overline{[\mathrm{chl}]} = .37\ \mathrm{X}^{.79}$	.88	.257	67	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = .74\mathrm{X}^{.89}$	.89	.284	45
Nordic.	$\overline{[\mathrm{chl}]} = .13\ \mathrm{X}^{1.03}$	.82	.329	13	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = .47\mathrm{X}^{1.00}$	.77	.373	13
Alpine.	$\overline{[\mathrm{chl}]} = .47\ \mathrm{X}^{.78}$	.94	.189	12	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = .83\mathrm{X}^{.92}$	.96	.191	11
Shallow Lakes & reservoirs	$\overline{[\mathrm{chl}]} = .54\ \mathrm{X}^{.72}$	.87	.238	22	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = .77\mathrm{X}^{.86}$	.88	.276	21
U.S.A	$\overline{[\mathrm{chl}]} = .39\mathrm{X}^{.79}$	.89	.261	20	–	–	–	–
Nordic Report . . .	$\overline{[\mathrm{chl}]} = .51\ \mathrm{X}^{.81**}$	.83	–	24	–	–	–	–
Alpine Report. . . .	$\overline{[\mathrm{chl}]} = .60\ \mathrm{X}^{.68}$	.90	.224	27	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = 1.12\mathrm{X}^{.84}$	.93	.230	26
Alpine Report. . . .	$\overline{[\mathrm{chl}]} = .50\ \mathrm{X}^{.73***}$	.90	.245	28	$\left[\begin{smallmatrix}\mathrm{max}\\ \mathrm{chl}\end{smallmatrix}\right] = .93\mathrm{X}^{.89***}$	.93	.230	26
S.L. & R. Report. .	$\overline{[\mathrm{chl}]} = .43\ \mathrm{X}^{.88}$	–	.414	–	–	–	–	–
U.S.A. Report. . . .	$\overline{[\mathrm{chl}]} = .55\ \mathrm{X}^{.76}$	–	–	45	–	–	–	–

* r = correlation coefficient, SE = standard error of estimate, n = number of data points
** summer chlorophyll
*** orthogonal equation.

programme. The relationship is, rather, due to the high correlation between nitrogen and phosphorus.

6.4.3. Regressions between Secchi Disc Transparency and Phosphorus Loadings

Secchi disc transparency has been found to be related to flushing corrected phosphorus inflow concentrations as follows (Figure 6.17, Table 6.8):

$$\overline{[\mathrm{Sec}]} = 14.7\left[\overline{[\mathrm{P}]}_j/\left(1+\sqrt{\mathrm{T(w)}}\right)\right]^{-.39}$$

The limitations of this relationship are similar to those given in § 6.2.

The same relationship for the regional projects is given in Table 6.8. The regression equations are not significantly different at the P > .95 level. An attempt was made to use the reciprocal value of the Secchi disc transparency in the Alpine report, but this approach did not improve the relationship.

Figure 6.17

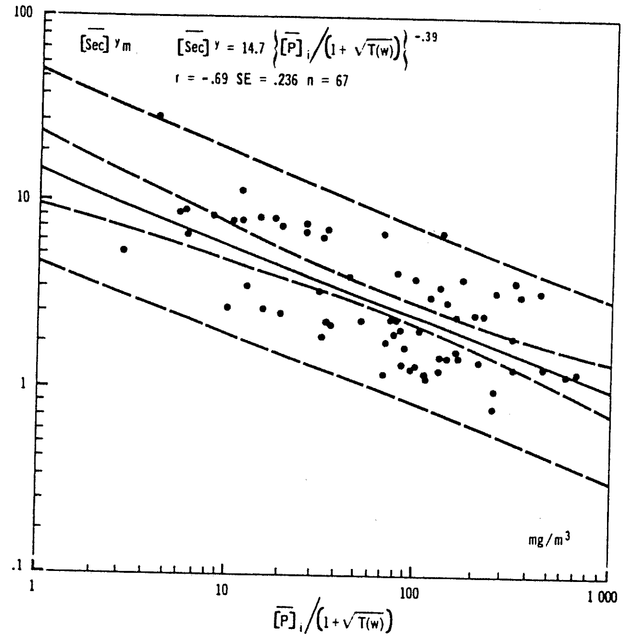


Table 6.8

REGRESSION EQUATIONS, RELATING SECCHI DISC TRANSPARENCY WITH FLUSHING CORRECTED PHOSPHORUS INFLOW CONCENTRATIONS, $[P](1+\sqrt{T(w)'}) = X$, FOR THE COMBINED, SCREENED OCDE DATA, REGIONAL PROJECTS AND REGIONAL REPORTS

(See Note in Appendix 3)*

Project	Equation	r	SE	n
Combined OECD data	$\overline{[Sec]}^y = 14.7\ X^{-.39}$	-.69	.237	67
Nordic.	–	–	–	–
Alpine.	$\overline{[Sec]}^y = 15.3\ X^{-.30}$	-.74	.171	18
Shallow Lakes & Reservoirs.	$\overline{[Sec]}^y = 8.47\ X^{-.26}$	-.55	.237	26
U.S.A..	$\overline{[Sec]}^y = 20.3\ X^{-.52}$	-.82	.196	22
Nordic Report	$\overline{[Sec]}^{su} = 12.7\ X^{-.50}$	-.78	–	–
Alpine Report.	$\overline{[Sec]}^{sp} = 9.31\ X^{-.16}$	-.53	.164	35
Alpine Report.	$\overline{[Sec]}^{su} = 8.41\ X^{-.21}$	-.68	.209	–
USA Report.	$\overline{[Sec]}^{**} = 8.4\ X^{-.36}$	–	–	–

* r = correlation coefficient, SE = standard error of estimate, n = number of data points.
** annual mean and summer values.

Figure 6.18

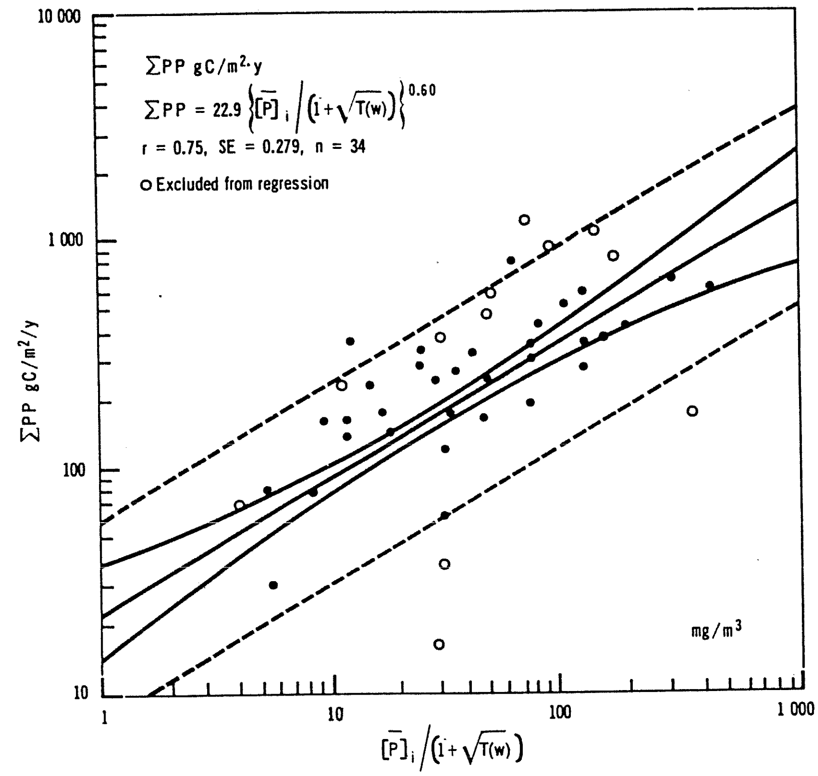
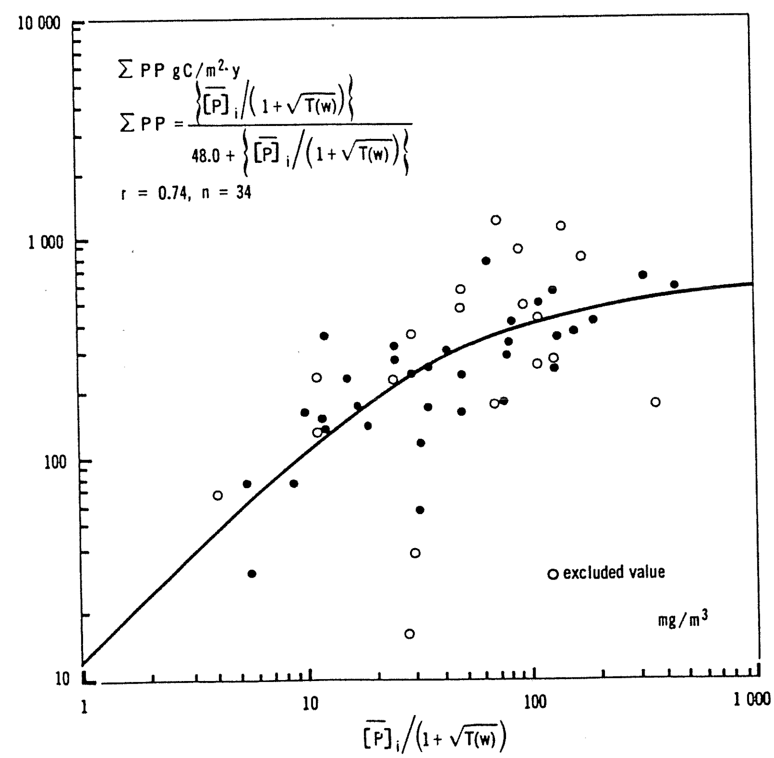


Figure 6.19



6.4.4. Yearly Primary Production from Phosphorus Loading

Yearly planktonic primary production has been found to be related to flushing corrected phosphorus inflow concentration as follows (Figures 6.18 and 6.19, § 6.3):

$$\Sigma PP = 22.9\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.60} \qquad r = .75,\ SE = .279,\ n = 34$$

and

$$\Sigma PP = 589\ \frac{\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]}{48.0+\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]} \qquad r = .74,\ n = 34$$

As found in §6.3, the relationship has a reasonably high predictive efficiency.

6.4.5. Regressions between Hypolimnetic Oxygen Depletion and Phosphorus Loadings

Regression equations of this sort have been calculated separately for the U.S. North American and the Nordic Projects respectively, whereby:

$$\Delta O_2,\ g\,O_2/m^2.day = .085\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.467}$$

and

$$\Delta O_2,\ g\,O_2/m^2.day = .115\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.67}$$

$$r = 0.63,\ n = 18$$

The derived equations are similar. Combining the data from the two projects, a preliminary equation has been established relating daily areal hypolimnetic oxygen depletion (g O_2/m^2.day) to the standard flushing corrected inflow concentration of phosphorus as

$$\Delta O_2 \approx 0.1\left[[\overline{P}]_j/\left(1+\sqrt{T(w)'}\right)\right]^{.55}$$

It is of interest to note that the exponents of the regression equations of the hypolimnetic oxygen depletion and yearly primary production versus flushing corrected total phosphorus inflow concentration, .55 and .60 (cf. § 6.4.4.) respectively, are very similar. This similarity suggests a close link between these relationships.

Considerable further work is required, however, to substantiate the hypolimnetic oxygen depletion-phosphorus loading relationship, and to test alternative relationships. Uncertainty exists in the definition of oxygen depletion, in relation to how the areal oxygen depletion rate reflects the transfer of the organic matter produced in the epilimnion into the hypolimnion. In the case of deep lakes in Italy where the deep waters are not mixed and oxygenated every year, the interpretation of the hypolimnatic oxygen depletion rates pose special problems. Also, the modifying effect of the volume of the hypolimnion (mean depth of the hypolimnion) could not be evaluated from the information presently available. The possibility of establishing a reliable relationship between phosphorus loading and hypolimnetic oxygen depletion rates will be important for predictive and lake management purposes.

Vollenweider, R. and J. Kerekes, 1982. Eutrophication Waters. Monitoring, assessement and Control, Organic for Economic Co-operation and Development. Paris, 154 p.

7. TROPHIC TERMINOLOGY

Preamble

Contrary to the aim of the previous section which attempts to establish quantitative relationships between parameters on which the trophic characteristics of lakes depend, section is intended to establish a bridge to classic limnological terminology which is qualitative in nature.

In the OECD programme the need for a clearer definition of the often vaguely used terminology (oligotrophic, mesotrophic, eutrophic, and their boundary categories ultra-oligotrophic, and hypertrophic, respectively has been recognised. Therefore, an attempts has been made to elaborate the available data with respect to total phosphorus, total nitrogen, average and peak chlorophyll, and Secchi disc transparency in such a was as to attach quantitative characteristics to the qualitatively defined trophic categories. From this, two alternative systems evolved, one which considers fixed reference boundaries, and the other which is opened.

7.1. FIXED BOUNDARY SYSTEM

The fixed boundary system is based on best judgement as to the transition between two neighbouring categories with respect to each parameter specified. These transition values are compromise values selected in such a way that in accounting for the regression relationships resulting from the OECD data elaboration, the parameter values are approximatively interchangeables (Table 7.1).

Table 7.1

PROPOSED BOUNDARY VALUES FOR TROPHIC CATEGORIES
(fixed boundary system)

Trophic Category	$\overline{[P]}_\lambda$	$\overline{[chl]}$	$\begin{bmatrix} max \\ chl \end{bmatrix}$	$\overline{[Sec]}^y$	$\begin{bmatrix} min \\ Sec \end{bmatrix}^y$
	mg/m^3			m	
Ultra-oligotrophic	$\leqslant$ 4.0	$\leqslant$ 1.0	$\leqslant$ 2.5	$\geqslant$ 12.0	$\geqslant$ 6.0
Oligotrophic.	$\leqslant$ 10.0	$\leqslant$ 2.5	$\leqslant$ 8.0	$\geqslant$ 6.0	$\geqslant$ 3.0
Mesotrophic.	10 - 35	2.5 - 8	8 - 25	6 - 3	3 - 1.5
Eutrophic	35 -100	8 - 25	25 - 75	3 - 1.5	1.5 - 0.7
Hypertrophic	$\geqslant$ 100	$\geqslant$ 25	$\geqslant$ 75	$\leqslant$ 1.5	$\leqslant$ 0.7

In this system, a certain arbitrariness is unavoidable, and the danger exists that the respective categorisation is rigidly applied. To avoid this, judgement about allocation of a given lake to a trophic category should not be based on only one or two parameters but on the total information.

The advantage of a fixed boundary system is its easy application by managers and technical personnel with only limited limnological training. In particular, it is apt to prevent gross misuse of the trophic terminology, which has often happened in the past.

7.2. OPEN BOUNDARY SYSTEM

In contrast to the fixed boundary system, the open boundary system is more flexible in application, and "outliers" may be more easily related to it. The system is based on group mean values and standard deviation for each parameter in question. These values have been obtained by tabulating the reported quantities into a 4 X 5 matrix in accordance with the subjective judgement of the author as to the trophic nature of the lake(s) investigated. Group means and standard deviations for each parameter selected have been calculated after log-transformation which was found to be necessary for normalisation (cf. Table 7.2).

The interrelationship between group mean maximum and minimum values, for phosphorus and chlorophyll, is presented in Figure 7.1.

With this procedure, the uncertainty in allocating a lake to a given category is taken into account and therefore, the *probabilistic aspect becomes an important judgement element in predictive application of the system.* In essence, it represents the qualified majority opinion of a large group of limnologists of how the trophic

Figure 7.1

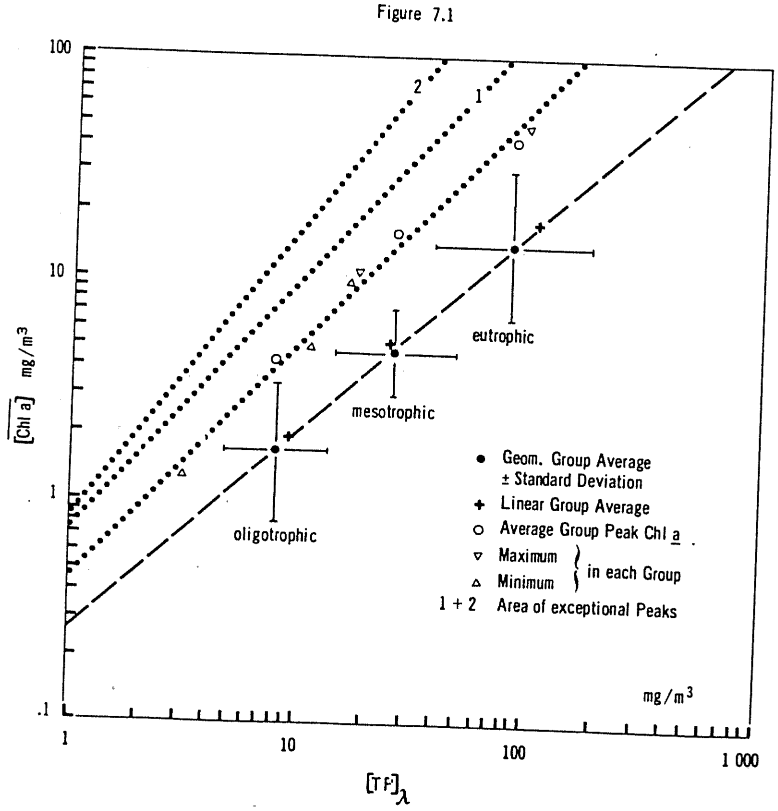


Table 7.2

PRELIMINARY CLASSIFICATION OF TROPHIC STATE
IN THE OECD EUTROPHICATION PROGRAMME
The geometric mean (based on log 10 transformation) was calculated after removing values < or > 2 SD obtained (where applicable) in the first calculation

Variable (Annual Mean Values)		Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Total Phosphorus mg/m^3		8.0	26.7	84.4	
	$\bar{x} \pm 1$ SD	4.85 – 13.3	14.5 – 49	48 – 189	
	$\bar{x} \pm 2$ SD	2.9 – 22.1	7.9 – 90.8	16.8 – 424	
	Range	3.0 – 17.7	10.9 – 95.6	16.2 – 386	750 – 1200
	n	21	19 (21)	71 (72)	2
Total Nitrogen mg/m^3	$\bar{x}$	661	753	1875	
	$\bar{x} \pm 1$ SD	371 – 1180	485 – 1170	861 – 4081	
	$\bar{x} \pm 2$ SD	208 – 2103	313 – 1816	395 – 8913	
	Range	307 – 1630	361 – 1387	393 – 6100	
	n	11	8	37 (38)	
Chlorophyll *a* mg/m^3	$\bar{x}$	1.7	4.7	14.3	
	$\bar{x} \pm 1$ SD	.8 – 3.4	3. – 7.4	6.7 – 31	
	$\bar{x} \pm 2$ SD	.4 – 7.1	1.9 – 11.6	3.1 – 66	
	Range	.3 – 4.5	3. – 11	2.7 – 78	100 – 150
	n	22	16 (17)	70 (72)	2
Chlorophyll *a* Peak Value mg/m^3	$\bar{x}$	4.2	16.1	42.6	
	$\bar{x} \pm 1$ SD	2.6 – 7.6	8.9 – 29	16.9 – 107	
	$\bar{x} \pm 2$ SD	1.5 – 13	4.9 – 52.5	6.7 – 270	
	Range	1.3 – 10.6	4.9 – 49.5	9.5 – 275	
	n	16	12	46	
Secchi Depth (m)	$\bar{x}$	9.9	4.2	2.45	
	$\bar{x} \pm 1$ SD	5.9 – 16.5	2.4 – 7.4	1.5 – 4.0	
	$\bar{x} \pm 2$ SD	3.6 – 27.5	1.4 – 13	.9 – 6.7	
	Range	5.4 – 28.3	1.5 – 8.1	.8 – 7.0	0.4 – 0.5
	n	13	20	70 (72)	

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

terminology is, and ought to be, applied in practice. Accordingly, two lakes with numerically similar characteristics (which is always only one part of a qualitatively oriented judgement) may appear in different (though neighbouring) categories. However, as a rule it may be assumed that a gross error in allocation is made if more than one of the parameters used to define the trophic nature of a lake deviates by more ± 2 standard deviations (Table 7.2) from the corresponding group means.

8. PREDICTION

Preamble

The following discussion is restricted to limnological situations which are within the range and types of lakes considered in the OECD programme for further discussion of the limitations imposed see § 9 . Here, it should be remembered that the standard relations resulting from the OECD study (see § 6) applied for predictive purposes, require careful evaluation of the situations and ranges for which these relations may, or may not be used. Gross errors in prediction could result if these aspects were ignored.

The numerical ranges over which the standard relations are valid, can be read out from the appropriate figures and tables.

In discussing application of the OECD results, three different topics are considered.

a) Quantitative Prediction.
b) Qualitative Prediction.
c) Prediction of Response Time.

8.1. QUANTITATIVE PREDICTION

The term "prediction" is used in this context with two different meanings.

a) prediction of the probability that a system (lake) will behave in certain respects if only partial information is available; (e.g. what is the most likely level of chlorophyll if phosphorus concentrations are known). The behaviour identified, then, is of *diagnostic value.* Conformity or non-conformity with expectation can be used to identify particular situations.

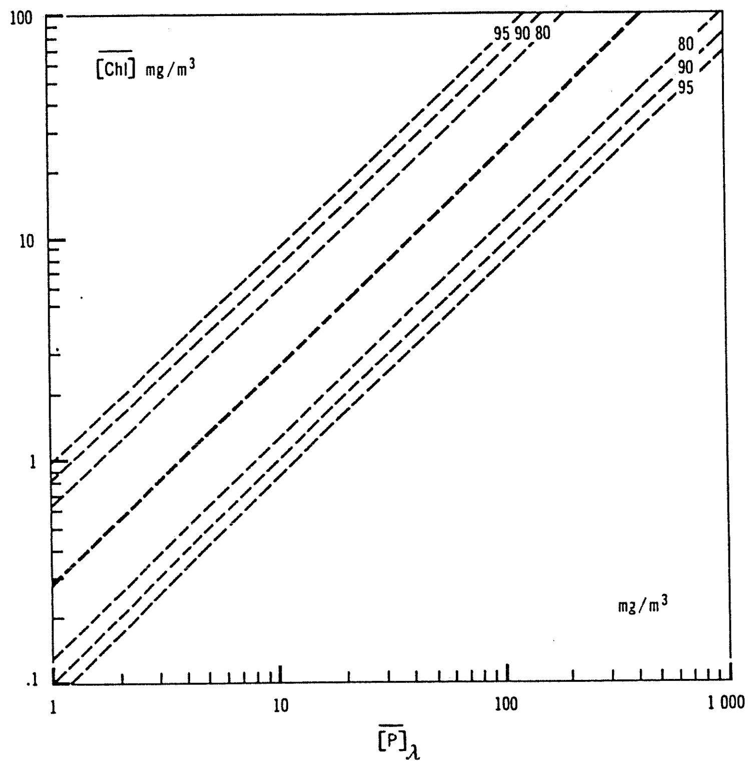
b) prediction of the range within which one can expect a system (lake) to react to changes imposed on it (e.g. nutrient load reduction). This aspect is of particular interest for *management,* and will be more extensively discussed in § 9.

The rationale for separating the prediction from the regression aspect relates to the need to account for *uncertainty.* Uncertainty analysis has not been considered here with the full care it requires.

Preliminary estimates of the level of incertainty have been made from simple statistics. For this diagrams are provided (Figures 8.1 - 8.5) in which the confidence intervals for the 80, 90 and 95% are drawn in addition to the regression line. The confidence levels relate directly to the scatter diagrams, and are derived from calculated standard errors of estimates (Figures 6.1, 6.2, 6.11, 6.13 and 6.14).

a) Because of the uncertainties and unspecified errors connected with the original data sets, the respective standard errors are high, even after elimination of questionable data points;

Figure 8.1



b) In the prediction of average lake concentration for phosphorus from flushing corrected loadings there is a smaller error than in the prediction of both chlorophyll from inlake concentrations, and chlorophyll from loading. However, the prediction errors for these two relationships are of the same order of magnitude. Therefore, the diagram should be used independently rather than sequentially (i.e. loading → concentration; concentration → chlorophyll).

c) In applying the diagram in either sense discussed above, it is suggested from the observed scattering of the original data points that for both purposes, the 80% confidence interval offers a reasonable working reference. Used in a diagnostic sense, e.g., this means that a new data point which lies within the 80% confidence limit, can be accepted as satisfactory; if the data point lies between the 90 and 95% confidence limit, it is possible that the data point does not belong to the population (i.e. it represents a particular situation) or that the basic data are wrong. If the data point lies outside the 95% confidence interval, it must be assumed that either it represents a non-conforming situation, or the basic information is wrong. If the basic information is proven correct, yet the lake represents a non-conforming, special case, it is important to analyse the reasons for the aberrant behaviour (see examples in § 5.1 and 6.1).

The specific use of these diagrams for practical lake management will be illustrated with selected examples in § 9.4.

Figure 8.2

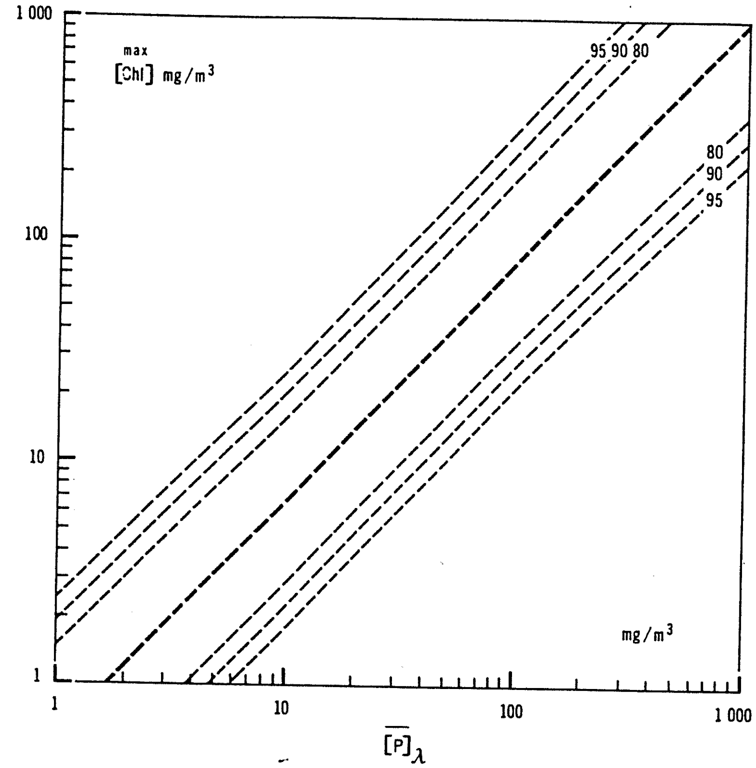


Figure 8.3

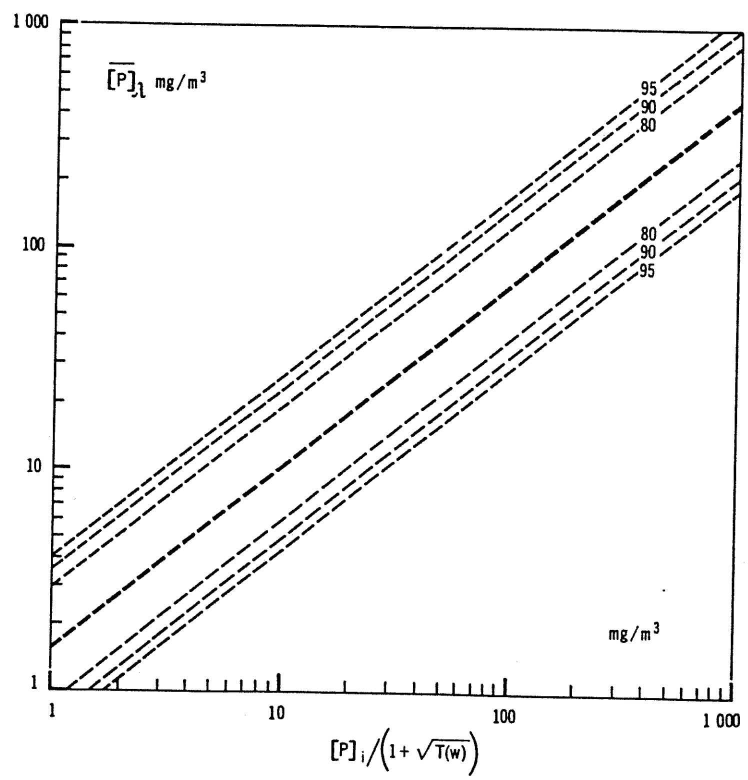


Figure 8.4

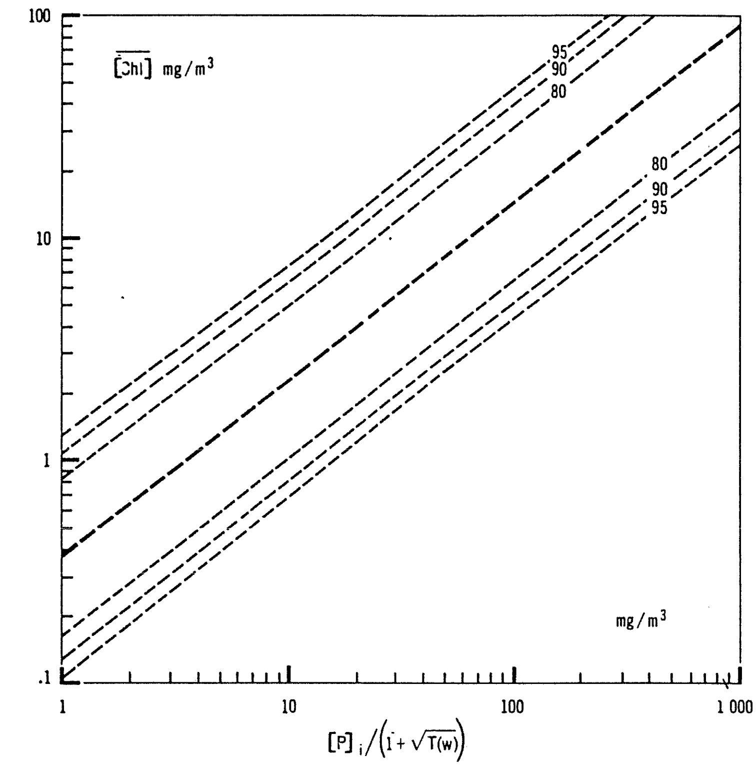
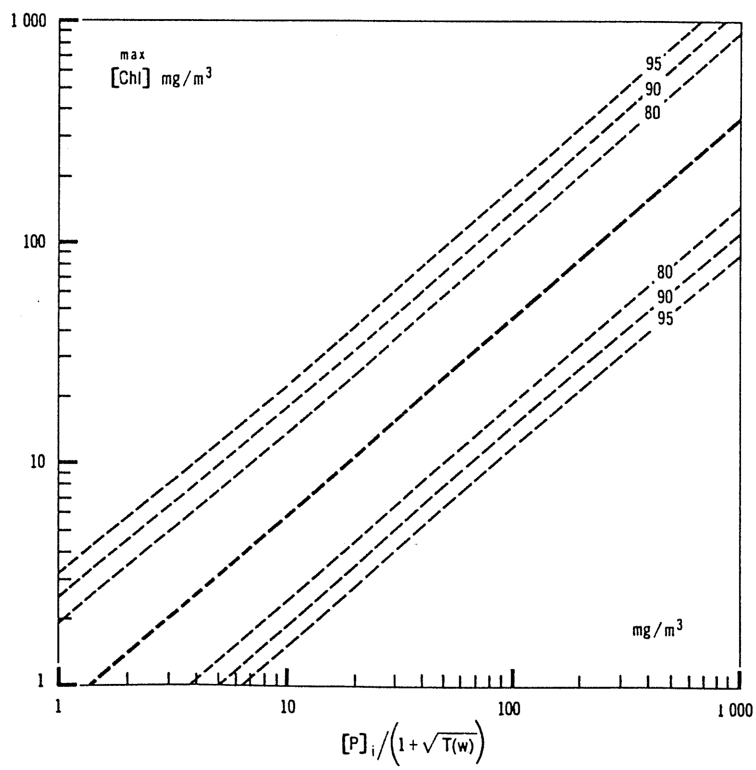


Figure 8.5



8.2. QUALITATIVE PREDICTION

For prediction of the qualitative characteristics (trophic categories), the results reported in § 7 can be interpreted in terms of probabilities. Basically, the respective prediction refers to an expected probability distribution for the classification of a given lake as oligo-, meso- eu- or hypertrophic, by a large number of limnologists, if for that lake quantitative data relative to the listed parameters is available. It is therefore not an absolutely unbiased way of classification. Nonetheless, objective reasons exists for the uncertainty of classifying a given lake in different catagories by two or more authors, depending on the management of that body of water.

Lake management problems can be related to trophic categories by considering the degree of impairment of use connected with each trophic category. For most water-use aspects impairment is minimal for oligotrophic lakes and highest for eutrophic and hypertrophic lakes while mesotrophic lakes occupy an intermediate position (see also Table 9.2).

Another factor is that average conditions, expressed by "average nutrient concentrations", "average biomass values", "average transparency", etc., do not necessarily express the degree of variability, particularly with regard to peak levels, frequency of their occurence, and their qualitative nature (type of phytoplankton). From the management view point, such situations and their frequency are as important as average conditions.

For this reason, prediction uncertainties must be accounted for. This can be achieved by reinterpreting the summary values listed in Table 7.2 in terms of classification probabilities. The resulting probability distribution is given in Figures 8.6 to 8.9 for the main components: average lake phosphorus, average and peak chlorophyll concentrations and average yearly Secchi disc transparency.

The five category system includes an ultra-oligotrophic and hypertrophic category in addition to the oligo-, meso-, and eutrophic categories. This enlargement is based on the fact that the group mean values of the three main categories (namely oligo-, meso- and euphotic) are separated by a factor of 3.

This system permits a prediction which is closer to reality; e.g., with regard to total phosphorus, a 10 mg/m^3 value has a probability distribution of 10% for ultra-oligotrophic, 63% for oligotrophic, 26% for mesotrophic, 1% for eutrophic and 0% for hypertrophic conditions (Figure 8.6).

For management, the following considerations are pertinent:

a) The example selected explains why exceptional cases like Lake Mjøsa with 10 mg P/m^3 must be expected (see Nordic Report).

b) For certain purposes, e.g. drinking water reservoirs, the objective may have to be more stingent than 10 mg/m^3 so that the probability that the water is preserved in pristine condition is sufficiently high.

c) In relation to hypertrophic conditions (as e.g. in Dutch waters), the diagram indicates that reduction of phosphorus would, at least, increase the probability of obtaining more acceptable eutrophic, or even mesotrophic conditions, although the return to oligotrophic conditions may be unattainable.

8.3. TIME RESPONSE

Little knowledge exists of response times of bodies of water after initiation of external nutrient loading reduction. Preliminary estimates of the expected response, for a given lake, can be made from equation 14 in Appendix 2, and Figure 8.10 provides a simple diagram for this purpose. The response time is calculated as a

Figure 8.6

PROBABILITY DISTRIBUTION FOR TROPHIC CATEGORIES

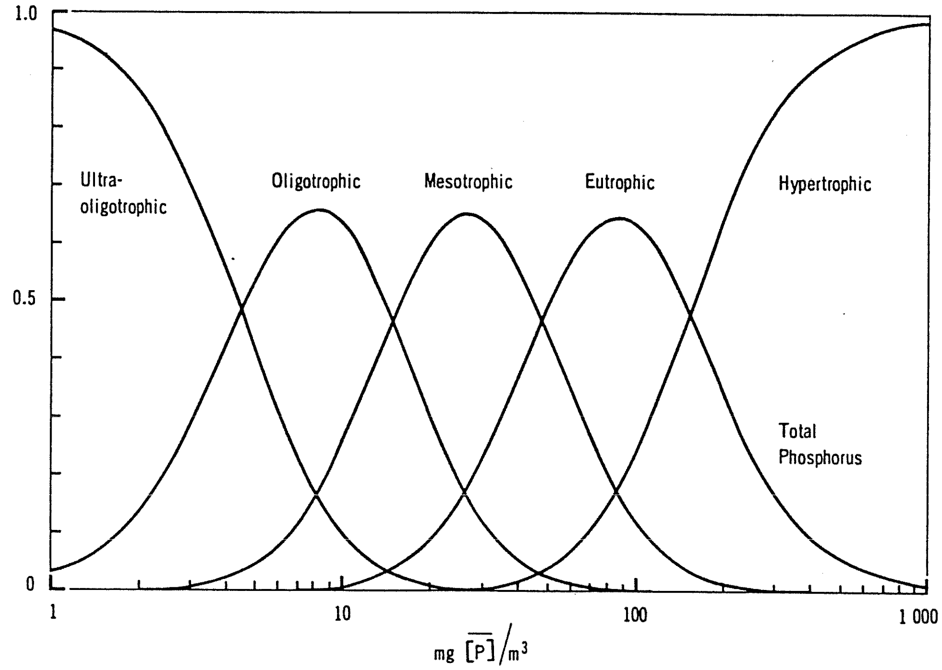


Figure 8.7

PROBABILITY DISTRIBUTION FOR TROPHIC CATEGORIES

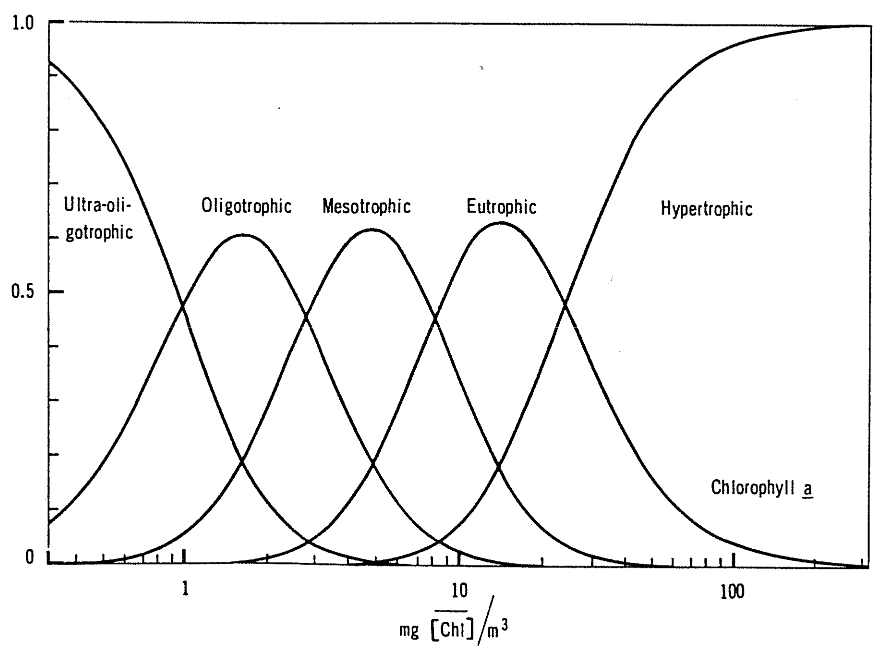


Figure 8.8
PROBABILITY DISTRIBUTION FOR TROPHIC CATEGORIES

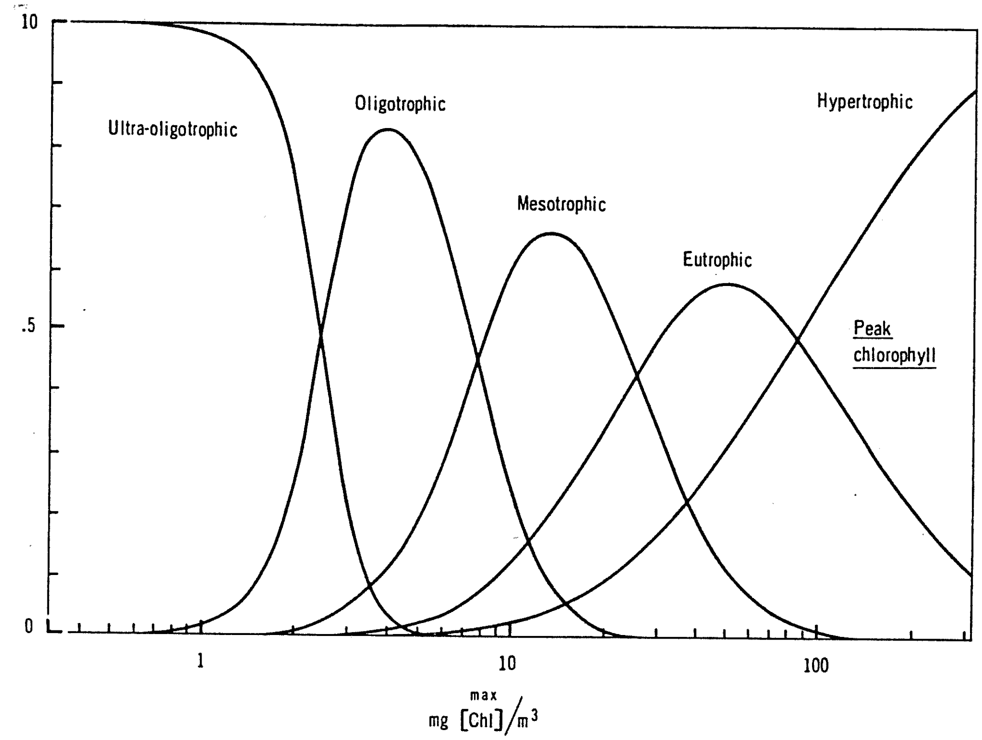
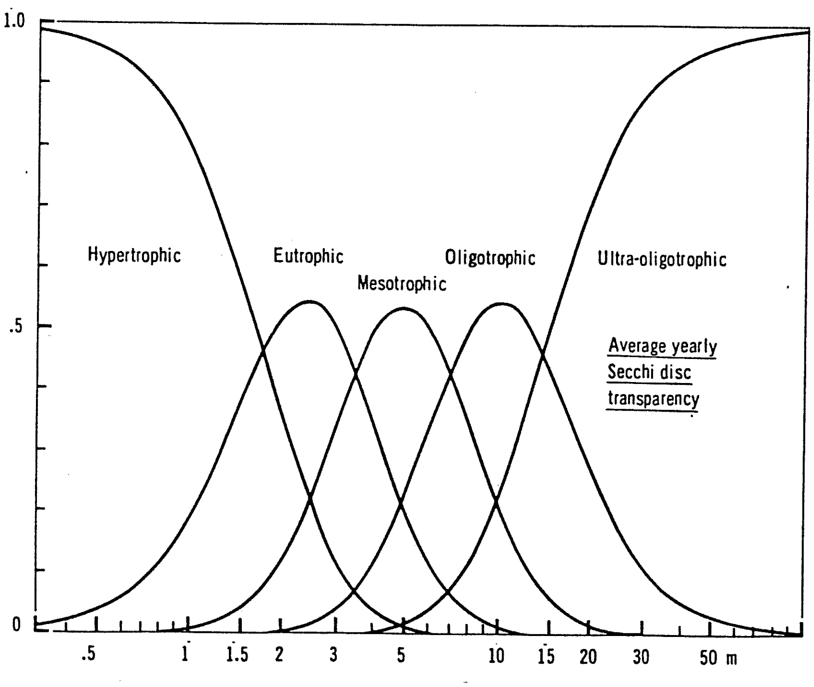


Figure 8.9
PROBABILITY DISTRIBUTION FOR TROPHIC CATEGORIES



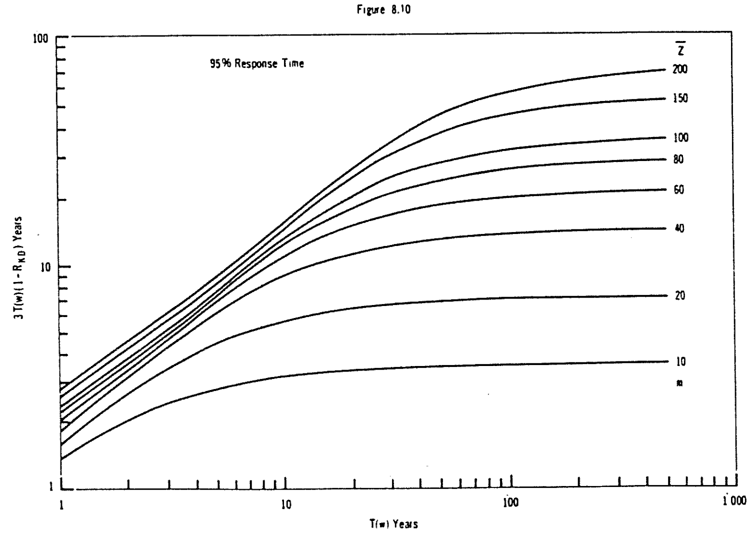
function of the theoretical water residence time and lake mean depth assuming that phosphorus is the key factor, and that the phosphorus retention coefficient is close to the Kirchner-Dillon relationship (see equation 9, Appendix 2).

From the diagram or from equation 14a, it would follow e.g., that a lake with a mean depth of 10 m, and a water residence time of 5 years (actual phosphorus retention coefficient approximatively 0.8), the expected 95% response time to reach a new equilibrium would be about three years.

The time response for lake recovery is sometimes estimated by using the phosphorus residence time. A new steady state phosphorus concentration would be approached exponentially as a function of phosphorus residence time after a reduction in phosphorus load (Sonzongni et al., 1976). As with the water residence time, 95% of the expected change in the mean phosphorus concentration, following external phosphorus load reduction, will be reached in a time period equal to the phosphorus residence time. The phosphorus residence time can be calculated by dividing the lake phosphorus concentration (mg/m^3) by the specific volumetric loading ($mg/m^3.y$) or by dividing the annual mean phosphorus content (gP) by the phosphorus input (g P.y). Since phosphorus is a non-conservative element, the phosphorus residence time is shorter than the water residence time. However, the relationship between phosphorus and water residence time is not linear. Therefore, with the information now available, it is possible to estimate the response time as a function of the water residence time and mean depth (Figure 8.10).

Difficulties may arise in estimating the response time when the internal loading component is high. Prediction of changes in the internal load component after reduction of the external load, is still difficult, particularly for lakes with a long history of eutrophication. Conversely, the little experience gained of lakes with a short history of eutrophication would indicate that for such lakes the actual response time is close to that expected.

Figure 8.10



9. APPLICATION OF OECD RESULTS TO WATER MANAGEMENT

9.1. INTRODUCTION

The waterbodies considered in the OECD Eutrophication Programme have a wide range of features with respect to morphometry, hydrology, nutrient load, trophic state and geographical location (see § 3.3.). Several lakes are in near pristine condition receiving only natural nutrient load, some have only been exposed to increased levels of man-made, cultural nutrient loads in recent years, while others have a long history of man-made eutrophication. Several lakes were submitted by nutrient load control either before or during the OECD Programme. The relationships between nutrient load, nutrient concentration and trophic response were analysed in the OECD lakes, and simple empirical formulae, derived from mass balance equations, were tested for predictive purposes. The OECD data has shown, that these empirical formulae can be used to predict trophic response to phosphorus loads and to estimate the necessary reduction in phosphorus load to achieve and maintain a certain trophic response (see § 8).

It can be concluded from the OECD Programme that increased phosphorus load in waterbodies results in a predictable increasing trophic response and the reduction of phosphorus load would elicit a predictable decrease in trophic response. The results of the Programme can be applied for water management purposes to control eutrophication in inland waterbodies.

Natural limnological conditions vary considerably among countries and also among different regions, particularly the larger countries. Consequently, the water quality objectives would differ in each country, taking local conditions and expectations into account. In the absence of human activities, the nutrient load and the trophic response in waterbodies are determined by the natural fertility of soils on the drainage basin which in turn depends on the geology and the climate of the area in question. Ideally, the objective of lake management should be to maintain or restore waterbodies to their natural state determined by the basic natural nutrient load of the area in question (e.g. free from human activities). In practice, this is not always possible.

The purpose of lake management is to control the "cultural" man-made nutrient contribution in order to restore to, or maintain waterbodies at the desired trophic state. The question of what is the desired trophic state of a waterbody is a management policy decision which would depend on the intended use of the water and the conditions and expectations in each country. The results of the OECD Programme provide lake management with the necessary tools to determine the phosphorus load reduction required to achieve a desired water quality as indicated by the expected trophic response. Once the required phosphorus load reduction is determined in a particular case, then an appropriate strategy and plan can be developed to bring about this reduction.

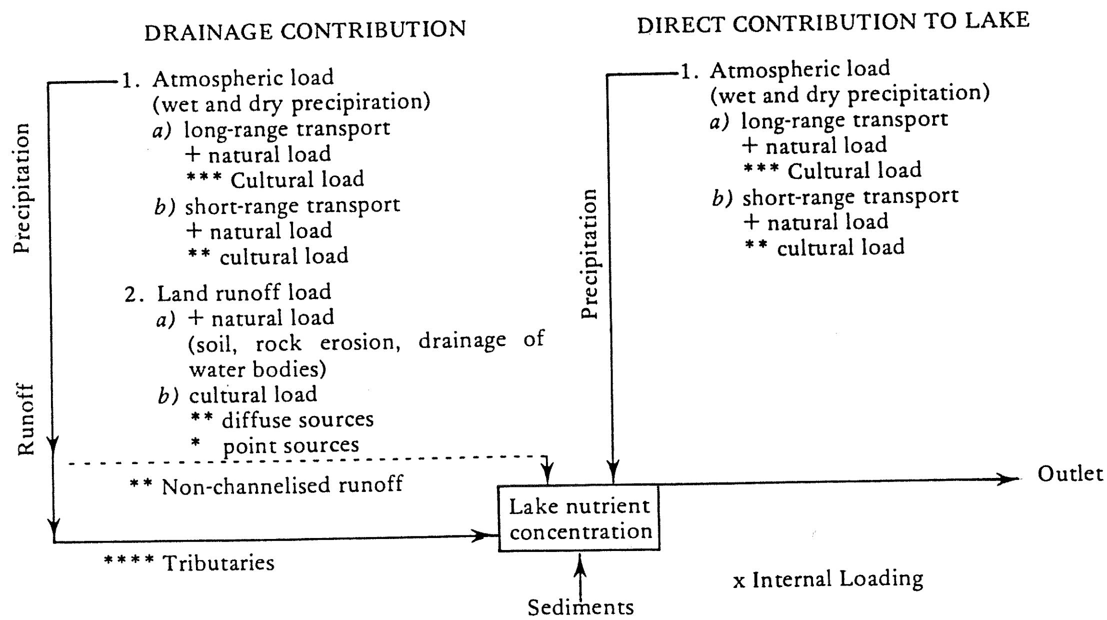
Since the desired water quality will vary among countries and geographical areas, the same specific phosphorus load may be interpreted differently in terms of water quality objectives in these areas. For example, a given phosphorus load may be totally unacceptable for many Alpine or Nordic lakes, yet the same load would be well below the natural phosphorus load in a lake surrounded by high natural soil fertility, and where this same specific load would be an unrealistic lake management objective.

9.2 NUTRIENT SOURCES

Water management considerations for eutrophication control should take into account all phosphorus sources that may reach a body of water (Figures 9.1, 9.2). These include the atmospheric load which contributes directly to the lake and the load from the drainage basin which is a combination of the atmospheric contribution and the runoff from the drainage basin. A more detailed list of the phosphorus sources is given in Table 9.1. From the combined sources, the fraction of phosphorus load which could be considered as the natural nutrient load (i.e, load not associated with human activities) should be identified. This is the basic natural load which under normal circumstances cannot be controlled and which determines the natural trophic response in waterbodies in the area in question. Once this natural load is known, the values obtained may be translated into export coefficients as per unit surface area of the drainage basin for drainage contribution and as per unit lake surface area for direct atmospheric contribution. The background, natural phosphorus load would then be the basis upon which realistic phosphorus control strategies and programmes could be

Figure 9.1

DIAGRAM SHOWING NUTRIENT SOURCES WHICH DETERMINE LAKE NUTRIENT CONCENTRATION. THE RELATIVE EASE OR DIFFICULTY TO CONTROL NUTRIENT LOAD FROM VARIOUS SOURCES IS INDICATED.



+	basic natural load, not possible to control under normal circumstances
*	easiest to control
**	difficulty to control
***	cannot be controlled locally
****	depending on the size of the triburaries, nutrient control can range from being relatively easy to being practically impossible.
x	internal loading, when present, expected to diminish with time after substantial reduction of external nutrient load.

Table 9.1

SOURCES OF PHOSPHORUS INPUTS THAT CONTRIBUTE TO THE PHOSPHORUS LOAD IN LAKES VIA THE TRIBUTARIES
Modified from Baret et al. 1977

1. **Point Sources**

   a) Effluents from sewered, populated areas
   – Human wastes;
   – Domestic wastes;
   – Food wastes;
   – Street runoff;
   – Industrial wastes.

   b) Industrial effluents discharges into watercourses.

   c) Effluents from water treatment plants.

   d) Street runoff (if separated from sewer system).

2. **Diffuse Sources**

   a) Effluents from non-sewered, populated areas
   – Septic tanks;
   – Domestic wastes;
   – Soil erosion.

   b) Effluents from cultivated land
   – Soil erosion;
   – Fertilizer losses;
   – Domestic animal excrements;
   – Organic plant wastes.

   c) Effluents from non-cultivated land
   – Soil erosion;
   – Organic plant wastes;
   – Wild animal excrements.

   d) Spring and natural waters.

   e) Reserves in bodies of water
   – Sediments
   – Fauna and flora;
   – Groundwater.

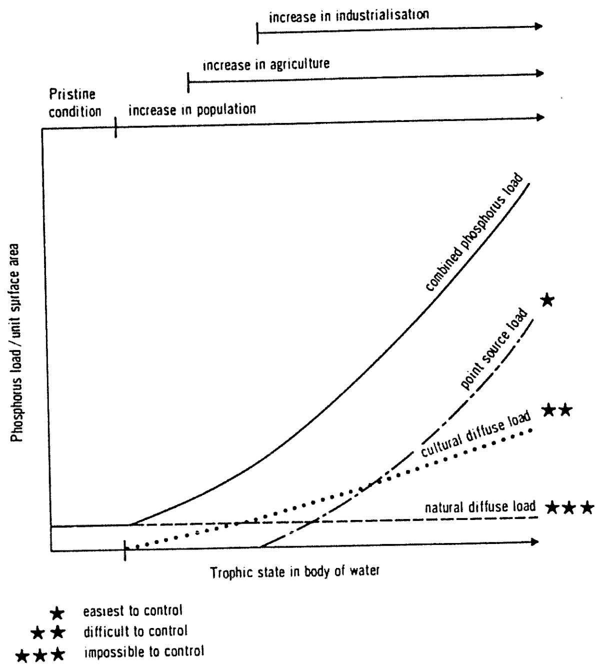
   f) Atmosphere
   – Wet precipitation;
   – Dry precipitation.

developed to achieve desired water quality and phosphorus load control objectives suitable for a particular country or geographical region. The remaining fraction of the phosphorus load which is the result of man's activities would be subjected to phosphorus load control.

The primary concern of water management is namely the development of water quality objectives for the multiple uses of water bodies. Once clear objectives for multiple uses are established, the problems for specialised users can be properly addressed.

In heavily populated, intensively cultivated areas, determination of the natural phosphorus load may be difficult. In such cases an educated judgement might be required to determine what constitutes the "natural" nutrient load for water management purposes.

Figure 9.2



Man-made sources of phosphorus may be divided into two categories, point sources and diffuse sources. Vollenweider (1968) gives a detailed review of both diffuse and point sources of nutrients. As the name implies, a point source of phosphorus is discharged at a single point into a tributary or directly into a lake. It may be industrial waste, sewage or storm sewer outlet, or a combination of these. Because of its definite location this phosphorus source is the easiest to control. In contrast, diffuse sources of phosphorus enter water courses in the runoff from non-sewed areas of the drainage basin. In the absence of defined discharges along the water course, the diffuse phosphorus sources are difficult to control. Any soluble substance containing phosphorus deposited on the drainage basin such as fertilisers, animal droppings, plant remains, waste products or eroded soil, from land use, is eventually carried into the water course and contributes to the diffuse sources of phosphorus.

With increased urbanisation and industrialisation, the relative importance of point source loadings of phosphorous progressively increases in relation to diffuse source loading, although the latter continues to increase in absolute terms (Figure 9.2).

The man-made phosphorus contribution can also be expressed in terms of export coefficients including most point source contributions. An extensive, and increasingly reliable literature of phosphorus export coefficients exists today for several geographical areas (Baret et al. 1977; Likens et al., 1977; Harrison, 1978; Ryding and Forsberg, 1979). Rast and Lee (1978) used phosphorus export coefficients to check the reliability of the phosphorus loading data for the U.S. part of the OECD Eutrophication Programme. However, care must be taken in applying and extrapolating export coefficients to areas that are not sufficiently known.

9.3 WATER QUALITY OBJECTIVES

Once the basic natural phosphorus load for a given area is determined, the natural nutrient concentration with the associated trophic responses can be estimated in a particular body of water according to its hydrological characteristics, by using the appropriate empirical, nutrient load-trophic response formulae (see § 8.1). Thus, the nutrient concentration and the trophic response in a lake in the absence of man-made accelerated eutrophication can be estimated. Knowledge of nutrient concentration and trophic response would be the ultimate objective of water management. This objective should be considered as a reference in the sense that it determines the lowest possible trophic condition that can be achieved in a water body and under normal circonstances, the nutrient concentration cannot be reduced below that level. However, socio-economic considerations and the intended use of the water would often prevent this ultimate water quality objective being achieved and hence a less stringent, more feasible water quality objective could be set. The knowledge of a natural nutrient load, however, should form the basis of water management considerations because it gives a solid foundation on which to set water quality objectives and it gives a proper perspective for eutrophication control.

Excessive supply of phosphorus in water is the cause of eutrophication, yet phosphorus in itself does not interfere with the normal use of water. The trophic responses caused by the nutrient enrichment such as high algal biomass, dense growth of macrophytes, reduced transparency, and reduced hypolimnetic or oxygen concentrations under ice cover are the symptoms of eutrophication which do interfere with normal use of the water. Therefore, water management should consider water quality objectives in terms of trophic response objectives, with regard to the intended use of the water. The trophic response objectives then can be translated into specific phosphorus load equivalents using the empirical nutrient load-trophic response formulae, upon which definite phosphorus load reduction objectives can be set.

When the notion of the "acceptable" and "excessive" loading boundary (10 and 20 mg $[P]_\lambda$ respectively) was introduced by Vollenweider (1968), and thus it first became possible to quantify the phosphorus loading relationship, this was based on the best limnological evidence available deduced from very limited data. It was an attempt to provide a starting point, from which to develop a trophic status index with respect to nutrient loading and trophic response, which could be tested and further refined. Since actual numbers were attached to the boundary line conditions between trophic categories, these numbers often became targets for eutrophication control. Because of the obvious connotations of the terminology of the boundary line definition, a lake with "excessive" or "unacceptable" loading was considered undesirable. It was often forgotten that in regions with fertile soils, water bodies exist in pristine conditions which are in fact, highly eutrophic and receive "excessive" natural nutrient load, which would put them well above the 20 mg $[P]_\lambda$ boundary line on a nutrient load-hydraulic load plot. In the original context, the denotation of "acceptable" and "excessive" loading boundaries were used to emphasise the effect of accelerated, man-made eutrophication on historically oligotrophic lakes where increased productivity and its consequences (i.e., algal blooms, low hypolimnetic oxygen concentration, loss of salmonoid fish, drinking water supply problems) implied a reduction in water quality. Water quality objectives relate to the intended use of the water. Therefore, the adjectives "good" or "poor" applied to water quality are meaning less, without reference to the intended water use. If this is to produce water fowl, the "good" quality water is a highly eutrophic one and an "excessive" loading, in the original sense, is acceptable in this case. Indeed, in certain parts of Atlantic Canada, deliberate management techniques are used to maintain or create eutrophic conditions in some shallow water bodies for the purpose of water fowl production to overcome the effects of "excessively" low (from the point of view of water fowl production) natural nutrient supply.

In areas of high soil fertility (e.g., The Netherlands, Prairie Provinces and States in North America) some naturally eutrophic waters become highly eutrophic as a result of excessive fertilisation. Under such conditions, the objective of lake management would be to maintain a moderately eutrophic condition, which is quite acceptable for many intended uses, particularly to the inhabitants of the area, in contrast to hyper-eutrophic waters which interfere with most uses.

Another problem area of water management is that of large lakes where local problems can develop due to increased nutrient load from smaller tributaries and sewage inlets. The lake as a whole may remain unaffected because the nutrient load of the tributary or sewage inlet is insignificant in relation to the whole lake which remains oligotrophic as an entity (e.g., low algal biomass in open water, high oxygen content in the hypolimnion). Yet small bays, and stretches of shoreline and the metalimnion near nutrient point sources could develop symptoms of excessive eutrophication. The total nutrient load may remain low enough to maintain an overall oligotrophic condition, viewing the lake as a mixed reactor, yet it may be necessary to intervene to reduce nutrient load from certain point sources to prevent or alleviate localised symptoms of undesirable eutrophication (e.g. Lake Mjøsa, Nordic Project).

It is a responsibility of lake management to decide what the quality objectives are for a water body as a function of its intended use. Based on that decision, a nutrient load control strategy could be developed. These objectives may vary considerably between different regions, depending on local conditions and on the expectation of the population concerning water quality. Depending on the intended use (e.g. drinking water, industrial water, power plant cooling water, recreation, multiple use, etc.), the stringency of objectives could be set at various levels, considering algal biomass (chlorophyll *a*) as a trophic response indicator, i.e. :

a) Mean algal biomass (annual, ice-free season, summer) to be kept below a certain level;
b) Mean annual peak algal biomass to be kept below a certain level;
c) Exceptional (highest possible) annual peak biomass to be kept below a certain level.

Considering the various trophic classes for the purposes of multiple use (disregarding uses which require eutrophic water such as certain fish ponds and water fowl production), oligotrophic lakes would create no problems, mesotrophic lakes would create some problems and eutrophic lakes would pose many problems for various users (Table 9.2).

The prediction uncertainty and the risks involved in the implementation of a phosphorus control measure undoubtedly creates uneasiness in the mind of many users (see § 8). However, as an increasing number of investigators begin to think in these terms and publish their findings, the task should become easier. The programme used mathematical and statistical tools to explain and predict lake behaviour, but it was built on limnological and not mathematical principles.

It was emphasied throughout the OECD Programme and in this report that the empirical formulae used to describe the nutrient load-trophic response relationships are based on statistical approximations. Therefore, accounting for uncertainty in formula prediction gives a better approximation to reality because it can estimate the probability that a certain water quality objective can be met. Thus, a measure of the probability of achieving an expected trophic response objective can be realistically considered by water management.

It is essential that water management uses limnological expertise both in the planning and implementation phases of any eutrophication control programme. Simplistic, blind application of the results of the OECD Programme should be avoided. In certain circumstances thorough limnological understanding may lead to alternative solutions for the control of accelerated eutrophication making costly phosphorus control intervention unnecessary.

The Queen Elizabeth II Reservoir in the United Kingdom is an exemple of such a solution. This highly specialised water supply reservoir is considered as hypertrophic, based on its annual phosphorus concentration (892 mg $[PO_4\text{-}P]_\lambda/m^3$), yet an astonishingly low mean annual concentration of chlorophyll *a* (2 mg $[chla]/m^3$) is maintained by means of artificial water circulation, the presence of silt, and relatively short water residence time. Without the artificial water circulation this reservoir would rapidly turn into an algal bowl as is suggested by the observed annual maximum

Table 9.2

TROPHIC CHARACTERISATION OF LAKES IMPAIRMENT OF VARIOUS USES

Limnological characterisation	Oligotrophic	Mesotrophic	Eutrophic
General level of production.	low	medium	high
Biomass.	low	medium	high
Green and/or blue-green algae fractions	low	variable	high
Hypolimnetic oxygen content.	high	variable	low
Impairment of multi-purpose use of lake	little	variable	great

chlorophyll *a* concentration of 48 mg $\left[\begin{matrix}\text{max} \\ \text{chl}a\end{matrix}\right]$. Other examples are known in which conditions or measures other than control of phosphorus supply were sucessful into eliminating the nuisance symptoms of eutrophication (Shapiro, 1978). These successes, however, do not mean that phosphorus control measures could be avoided in the majority of cases but they do emphasise that full limnological understanding is essential when contemplating measures to control accelerated eutrophication.

9.4. USE OF OECD FORMULAE FOR ASSESSING EXTERNAL PHOSPHORUS LOADING REQUIREMENTS

In the following, some specific examples are given in relation to how to apply the results of the OECD study to practical lake management. Prior to this, it is of utmost importance, however, to keep in mind the limitations imposed by both the kind of programme conducted, and the results obtained.

It is recommended that the results of the programme are not applied, or extrapolated to geographical regions and limnological situations which differ grossly in their physiographic and climatic characteristics from those covered by the programme. This would exclude all tropical and most subtropical situations, as well as extreme northern and high mountain regions, and the marine environment (1).

Clearly, limitations also exist within the geographical regions and physiographic characteristics covered by the programme, since not all limnological conditions within these boundaries have been adequately considered. For typically meromictic lakes and lakes of special ionic characteristics, such as saline lakes, non-calcium-hydrocarbonate lakes, marl lakes and bog lakes, prediction should be undertaken with extreme care. The same cautionary note applies for very shallow lakes with prevailing macrophyte communities, for groundwater seepage lakes, and for all lakes with unusual physiographic and hydrographic characteristics.

Within the classes of lakes for which the study has been designed, special conditions can exist which, while not entirely invalidating prediction, will need careful evaluation. This applies primarily to predicting recovery time for lakes with a long history of eutrophication leading to a high organic content in sediments, severe hypolimnetic oxygen deficiency, high internal load, etc., or which have particular a fish stock, known to stir sediments and increase internal recycling of nutrients, for exemple carp.

1. Preliminary attempts have been made in the U.S. to extent the OECD approach to such environments.

During the execution of the OECD Programme, essential progress has been made in dynamic modelling. With such models of various combinations alternative measures can be evaluated. However, these techniques require a considerable amount of basic data, and the models have to be adjusted for each individual case.

Let us consider the following cases for which it is assumed that the above restrictions do not apply, and hence, that the formulae and diagrams given in this report can be used. Two basic objectives for control are examined:

a) to keep *average production* at a prefixed level; and
b) instead of average production to keep *peak value* below a prefixed level.

In either case these objectives may be more or less stringent, and furthermore, the specific control criteria will depend on whether the case in question is evaluated in more optimistic or more pessimistic terms. For an optimistic view one can assume that the lake, in terms of its trophic reaction, follows average standard conditions; for a more pessimistic view, one has to assume that the trophic reaction will be less than average.

These two control objectives, and their variations, are illustrated in the following examples whereby it is supposed that for the pessimistic evaluation the 80% level of uncertainty gives a reasonable reference when using Figures 8.1 to 8.5.

Example

Evaluation or required phosphorus load for a water body with an average water residence time of 9 years.

Objective A.— Average chlorophyll level to be kept at 3 mg/m³.

A. 1. Optimistic evaluation -

From Figure 8.4 it follows that:

$$\overline{[P]}_i/\left(1+\sqrt{T(w)'}\right)\Big] = 14;$$

this corresponds to an average inflow concentration of

$$\overline{[P]}_i = 14\left(1+\sqrt{9'}\right)\Big] = 56 \text{ mg/m}^3.$$

A.2. Pessimistic evaluation

Instead of considering the average regression, we use the upper 80% curve in Figure 8.4. For 3 mg this corresponds to 5.2. Accordingly, the permissible average inflow concentation would be

$$\overline{[P]}_j = 5.2\,(1+\sqrt{9}) = 21 \text{ mg/m}^3.$$

Objective B.— Peak chlorophyll should not exceed 15 mg/m³.

B. 1. Optimistic evaluation

For this Figure 8.5 gives:

$$\overline{[P]}_j/\left(1+\sqrt{T(w)'}\right)\Big] = 29$$

Accordingly, the permissible average inflow concentration should not exceed

$$\overline{[P]}_j = 29\,(1+\sqrt{9'}) = 116 \text{ mg/m}^3.$$

B.2. Pessimistic evaluation

The corresponding value in Figure 8.5 at the upper 80% uncertainty level is 11; accordingly, if the 15 mg level must not be exceeded, then the maximum permissible inflow concentrations must not exceed:

$$\overline{[P]}_j = 11\,(1+\sqrt{9'}) = 44 \text{ mg/m}^3.$$

The pessimistic situation, in fact, exists if the water body in its present condition already tends to be placed in the upper range on the $[chl]/[P]_\lambda$ diagram. A more optimistic view can be taken, if this is not the case. It is evident from the calculations presented that for the pessimistic situation, the control measures have to be substantially more rigorous than for the optimistic situation.

Accordingly, for Objective A, the total average yearly inflow concentration of phosphorus from all sources (considering source strength and total water discharge volume) should not exceed 50 to 60 mg P/m^3, and hopefully, should be less. To meet Objective B, on the other hand, the corresponding average inflow concentration should not exceed 110 to 120 mg P/m^3.

The tropic category diagrams, as well as the other diagrams (Figure 8.6 to 8.9), are an additional aid in evaluation procedures. With regard to case A, for example, one could expect the lake to average out at a $[P]_\lambda$ concentration of roughly 14. With this one could expect the following probability distribution of trophic characteristics for the lake at its new equilibrium: 59% mesotrophic, 32% oligotrophic, 7% eutrophic, and a residual probability for ultra-oligotrophic. Therefore, it is most likely that the lake will have mesotrophic characteristics with a slightly higher chance of being in a better condition.

If instead, the lake reacts less favourably with the same inflow control, then one might expect the lake concentration to remain as high as 25 mg/m^3. Therefore, the possibility of the lake remaining more oligotrophic would be reduced, and the lake would be more likely to become average mesotrophic. Conversely, if the more stringent objective is considered (i.e. $\overline{[P]}_j/(1+\sqrt{T(w)})] = 5.2$), then the expected average lake concentration would be about 6, or in the worse case, about 10.

With this, the chances of the lake becoming oligotrophic, or even better, increase dramatically even if the less favourable assumption is made.

Management will have to decide what is acceptable in terms of objectives and the practicality of reaching these objectives through control of the external load alone) or if additional or alternative control measures will have to be taken (see § 9.2).

Shortcut for Average Loading Objectives.

Figure 9.3 synthesises the standard OECD equations for the relationships between average inflow phosphorus concentration, expected average lake concentration and expected average chlorophyll concentration as a function of the average water residence time.

This diagram also gives approximative indications of the expected trophic category. As these categories are management oriented, they are slightly more stringently defined (i.e. approximately at the class midpoints) than are the categories used for diagnostic purposes. This provides a certain safety margin for the design of the loading objectives.

The diagram can be used for a rapid and crude evaluation of the required phosphorus load to achieve average preset conditions. For example, for a lake with an average water residence time of 10 years to reach an objective of 2 mg/m^3 average chlorophyll, the average phosphorus inflow concentration should be about 30 mg/m^3. To calculate the corresponding load, the following relationship can be used:

$$\text{Total Phosphorus Load} = \frac{\overline{[P]}_j}{T(w)} \,.\, \bar{z} \,.\, A_0 \,,$$

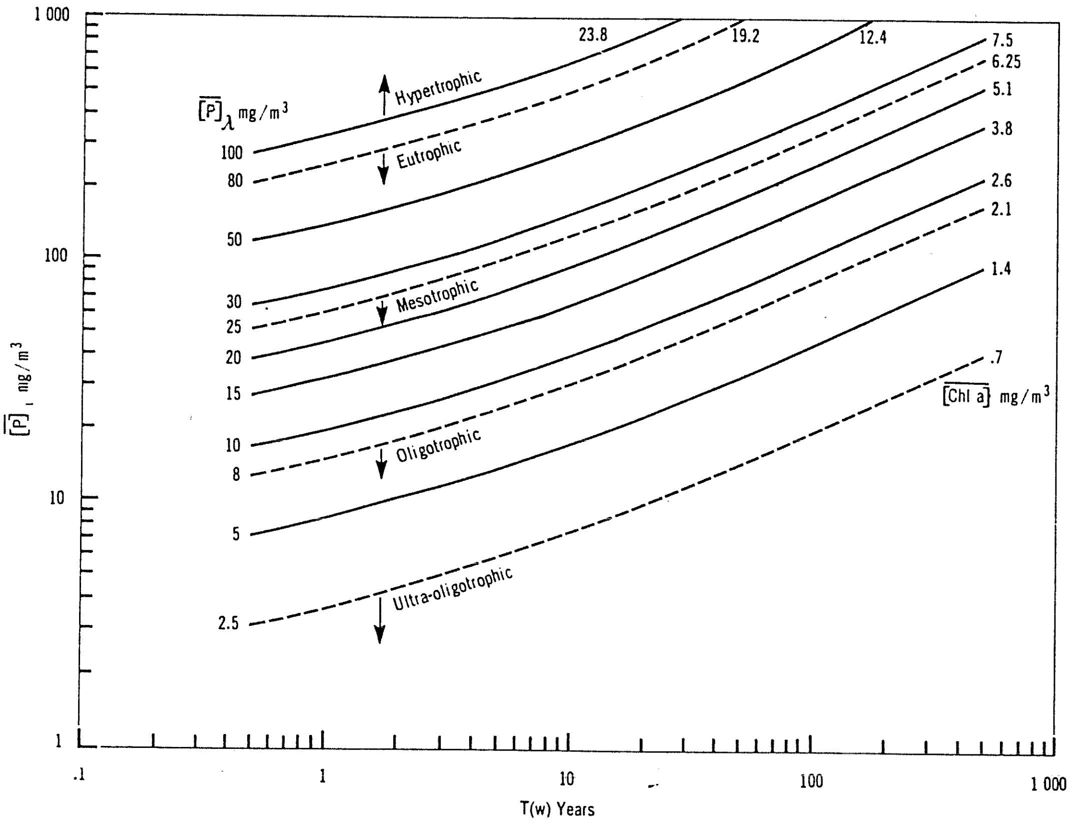
$\bar{z}$ and A_0 in meters and square meters. If the lake in question has a mean depth of 50 m, and a surface area of 200 km^2 (200.10^6 m^2), then the required load would be

$$\text{Total Load} = \frac{30 \times 50}{10} \;.\; 200 \;.\; 10^6 \text{ mg/y}$$

$$\text{(or)} = 30 \text{ tonnes.year,}$$

i.e the total load should be reduced to about 30 tonnes of phosphorus per year.

Figure 9.3



9.5. EVALUATION OF NEEDS FOR CONTROL MEASURES OTHER THAN EXTERNAL NUTRIENT CONTROL

The OECD Programme has not been designed to study and evaluate all possible eutrophication control measures which, in specific cases, may be applied alone or in various combinations with or without external nutrient control. These measures are not under review here, but a short summary of them is given in Chapter 10.

However, use can be made of the OECD results to evaluate, in principle, the need for control programmes and measures other than straightforward control of the external nutrient load. Indeed, external nutrient control in many cases is either impossible, or only partially attainable.

The OECD results, and their application as discussed in 9.4, indicate the level of external nutrient control necessary to achieve a prefixed water quality objective, and in this sense provide a yardstick for evaluating whether or not, with technologically and economically realistic nutrient reduction programmes alone, the desired water quality objectives are attainable. If the management information available indicates that external nutrient control alone would either be insufficient, or disproportionately costly, then alternative solutions may be proposed.

This is of importance in the sense that, in the past, various methods for eutrophication control were often undertaken on an ad hoc basis, and as a consequence in several cases, the control measures adopted failed to achieve the objective. Also, cases are known where external nutrient load reduction was insufficient, and therefore improvement is trophic conditions was only marginal as can now be seen in the results of the OECD Programme.

9.6. CASES WHICH DO NOT CONFORM WITH THE RESULTS OF THE OECD PROGRAMME

In several sections of this report, reference has been made to what have been termed "outliers". In view of the enormous variation of limnological situations which are either natural, or due to man's intervention, it would be impossible to deal with all cases in the way discussed above. Certain cases are of interest which – according to their general limnological conditions – should follow the OECD pattern but, in fact, do not. Here the OECD results and their application help to identify the existence of a specific situation which requires special attention and, as for many cases of this type, needs additional research to elucidate the reason(s) for the particular behaviour. Such reasons may be manifold, and quite different in the nature. They may be physiographical, hydrological, physical, chemical, biological, etc., but also result from industrial discharges of a particular kind. In many such cases nutrient control, whether external or internal, may not be the sole means by which to restore their conditions without intervention into their specific characteristics, if such intervention could ever be possible.

9.7. CONTROLLED FERTILISATION

The objective of the OECD Programme has been to provide guidelines for eutrophication control, i.e. restoration of water quality from excessive fertilisation to a level of lower and more acceptable trophic conditions. The following, therefore, is to some extent contrary to the above objective. Exceptionnaly, it might be desirable to *increase* the fertility of a water body e.g. for improving fisheries and/or sustaining appropriate conditions for water fowl. In this case, advantage can also be taken of the OECD results for evaluating what level of fertility would be desirable and compatible with such an objective.

Controlled fertility in this context could mean both restoration from excessive fertility to a lower level, (but not lower than a desirable), or artificially increased fertility (but not to above a desirable level), with appropriate dosage of nutrient supply.

10. EUTROPHICATION CONTROL STRATEGIES

10.1. PHOSPHORUS CONTROL STRATEGIES

The objective of water management is to restore and/or maintain bodies of water in a desired trophic state. This can be achieved by reducing and maintaining the total phosphorus load at a level which would result in a desired trophic response according to the nutrient load-trophic response relationship used in the OECD Programme. However, the development and the implementation of phosphorus load control measures is not purely a scientific and technical problem. Economic, legal and political considerations play a dominant role in these matters. In the case of large drainage basins (e.g. North American Great Lakes, Bodensee, Lac Léman), two or more countries are involved with several administrative levels of jurisdiction within each country. Successful phosphorus load control in such water bodies cannot be achieved without the coordinated cooperation of all the levels of jurisdiction concerned. This could mean treaties, agreements, standards for sewage effluents, special laws and proper enforcement. It is likely that municipalities, situated some distance from the receiving water body, might feel penalised if they are obliged to contribute to treatment cost since this may not be for their immediate benefit. Also, industries could feel that they have to carry an additional financial burden which is not imposed on other similar industries discharging directly into the ocean or into watercourses not affecting lakes. These are economic and political problems requiring political solutions based on economic and legal considerations which would vary considerably among different countries.

Accelerated man-made eutrophication and consequent environmental degradation result from increased nutrient loads caused by increased economic activity. Control of accelerated, undesirable eutrophication in water bodies requires a financial commitment to pay for both corrective and preventive phosphorus control measures. The problem of who should bear the cost has to be resolved before effective measures can be implemented.

For effective control of eutrophication, the control programme has to be a well-coordinated effort with the drainage basin viewed as one unit. In addition to present conditions, the projected population and economic growth of the basin must be considered. Furthermore, the relevant economic and legal implications should be properly addressed and resolved when control measures are planned. The control measures should be planned according to a clear set of priorities to prevent economic waste. This would mean that control measures are applied first to the phosphorus sources in which the greatest amount of biologically available phosphorus can be eliminated per unit cost. In large water systems it is essential to establish priorities in relation to *where to control phosphorus sources. For exam*ple, in the North American Great Lakes, the eutrophication problem exists in the Lower Lakes. Thus, phosphorus control measures in the Upper Lakes would contribute very little or nothing to improve water quality where the eutrophication problem is most severe (Chapra, 1979). Uncoordinated, piecemeal efforts to control eutrophication are economically wasteful and unlikely to succeed.

The measure of success of eutrophication control is the most economical achievement of the trophic response objective in a water body. The number of sewage treatment plants on a watershed does not necessarily demonstrate that sufficient efforts have been made to improve water quality (Ambühl, 1979). Indeed, fewer efficient plants treating carefully selected sources could produce more satisfactory results at a lower investment and operating cost per unit of biologically available phosphorus removed, than a large number of inefficient plants treating less important sources of phosphorus.

The conditions and the extent of the problems associated with phosphorus control measures would vary from case to case and different solutions might be sought for specific situations. A solution that might work in one case could be totally inappropriate in another because of limnological or economic conditions or because of the intended use of the water. To find the best solution for each case a sequence of priorities should be followed to assess the problems.

1) Establish a coordinative and administrative machinery to assess the problem, viewing the drainage basin as a unit for the purpose of eutrophication control. Coordinating bodies of this type already operate in the OECD countries, for example Commissions and Boards for the North American Great Lakes, Lac Léman, Bodensee, Wahnbach Reservoir). Such a body would recommend realistic and appropriate water quality objectives, recommend target load(s) for the desired trophic response, and establish priorities for phosphorus source control measures. It would also assess the financial requirements and identify legal problems that may have to be resolved before the corrective measures can be implemented. Once a thorough assessment of the situation is made, the coordinating body would be required to produce a management plan to implement the phosphorus control programme. The management plan should include at least the following points (modified after Annon., 1978):

   a) a timetable indicating programme priorities for the implementation of recommendations;
   b) agencies responsible for the ultimate implementation of programmes designed to fulfil the recommendations;
   c) formal agreements made to ensure inter- and intra-jurisdictional cooperation;
   d) sources of funding;
   e) water quality (trophic response) objectives to be achieved;
   f) estimated reduction in loadings to be achieved;
   g) priorities for phosphorus source controls;
   h) estimated costs of these reductions; and
   i) provision for public review.

   It should not be forgotten, however, that the fundamental problem to be resolved in lake restoration is of a biological nature and therefore skilled limnological expertise should always be included in the workings of commissions or boards in charge of lake restoration.

2) Assess the existing trophic conditions and the existing phosphorus load from all sources (bearing in mind the projected population and/or industrial growth in the basin). Establish a water quality objective (expected trophic response) taking into consideration the intended use of the water and the natural trophic conditions of the area.

3) Determine the amount of phosphorus load that has to be removed to meet the required trophic response objective. Depending on the size, shape and hydrological conditions of the water body, select the appropriate phosphorus load-trophic response formula upon which a specific phosphorus target load can be estimated. In large water bodies, and those of complex morphometry, the different parts may have to be treated separately and local effects such a shore conditions should be recognised.

4) Determine whether control of the point source load, which is the simplest to carry out, is sufficient to reach the specific target load. If the amount that can be controlled at point sources is not sufficient to meet the objectives, determine the percentage of the diffuse phosphorus load, which is usually more difficult to control, that should be controlled to meet the required phosphorus load reduction.
5) Asses realistically the total amount of phosphorus that can be expected to be removed from all sources. If this amount falls short of the required target, indicate the trophic response that could be achieved with the best possible efforts in phosphorus load reduction.
6) Assess the time required for recovery of the water body assuming that all the proposed phosphorus source control measures can be implemented (see § 8.3).
Depending on the nature of the case (e.g. long history of accelerated eutrophication, long water residence time) the restoration of a water body may take a considerable time.

In a water body which has been anoxic for a long period and where heavy internal loading exists, the time required for recovery would be much longer than that estimated from the prediction formula based on hydraulic residence time (e.g. Baldeggersee); (cf. Ahlgren, 1977; Ryding and Forsberg, 1977, Schindler et al. 1977). If, however, it has only been eutrophic for a few years, an almost immediate recovery can be expected after the reduction of the external phosphorus load (cf. Michalski and Conroy, 1973; Schindler, 1975). In situations where the lake has been eutrophied for longer periods, the recovery would be somewhat slower (cf. Edmondson, 1972; Larsen et al., 1975).

10.2. PHOSPHORUS CONTROL PRIORITIES

The selection of control priorities should be done by careful case by case evaluation of the existing problem(s) in the receiving water body considering its particular characteristics (e.g. morphometry, hydrology) and the intended use of the water.

Under normal circumstances, the first priority would be to control large point source domestic sewage inlets with a high percentage of biologically available phosphorus, where corrective measures could be undertaken most effectively. Depending on the purpose of the restoration programme, e.g. where the correction of local effects is stressed, the control of point sources that cause the local effects would be of highest priority. Sources that supply less total phosphorus or have a low biologically available percentage would be of lower priority.

Preventive measures that could produce appreciable reduction in phosphorus load should take very high priority. These measures may include the reduction, or replacement of phosphates in detergents. These measures are particularly important where some domestic sewage is discharged into a lake or where treatment facilities do not include chemical precipitation.

When priorities are established for phosphorus source control, it should be realised that effective control measures often require a coordinated combination of different approaches for the best results. For example, where tertiary treatment is contemplated, the separation of the storm sewer system from the domestic sewer system would greatly increase the efficiency of phosphorus load reduction. Treatment plants operate on the basis of a certain allowable phosphorus concentration in the effluent (e.g. < 1 mg P/1). Reduction of the quantity of water to be treated would lower the operational costs of the plants. On the other hand, treatment plants cannot normally handle exceptional storm runoffs. During such events, the plant cannot maintain the required effluent phosphorus concentration standards and large quantities of phosphorus may be discharged inadvertently from the plant, which defeats the purpose of the treatment effort.

In situations where point source phosphorus reduction cannot meet the target load objective, diffuse source load control should be given high priority. This could mean the modification of cultivation and land use practices. It should be recognised that remedial measures to control diffuse sources may imply additional costs, and the benefits resulting from these costs may not be felt directly by those who pay for them. In these instances, incentive schemes should be proposed so that individuals or agencies find it worthwhile to undertake the required measures.

It is essential to estimate realistically the cost of the phosphorus source control measures so that the control priorities may be established on sound economic bases. This would require a careful, case by case evaluation of the existing problem(s) in the receiving water body, taking into consideration its particular characteristics (e.g, morphometry, hydrology, etc.), the intended use of the water and the conditions existing in the drainage basin. After a certain initial effort, additional point source control could become more expensive than certain diffuse source control measures (Table 10.1). For example, the reduction of the phosphorus load by the reduction of waste water plant effluent concentration below 0.5 mg/l might be more costly than the same reduction in load by various diffuse source control measures. In the case of Lake Erie, good management of all agricultural land, avoiding excessive fertilisation, could reduce the phosphorus load by 10% at no extra cost (cf. rural nonpoint sources, Table 10.1). The implementation of a more advanced rural nonpoint source control (Level 2) which includes buffer strips, strip cropping, etc., would greatly increase the unit cost of phosphorus reduction to $64.3/kg, which is still less expensive than the $95.5/kg for the reduction of waste water treatment effluent concentration below 0.5 mg P/1. However, by increasing the rural nonpoint source control to a greater intensity (Level 3), the unit cost of phosphorus reduction would increase to $174.0/kg. Under such conditions the reduction of the sewage treatment effluent from 0.5 to 0.3 mg/l would be an attractive alternative. In addition to the high costs of such advanced agricultural practices in terms of phosphorus control, it should be kept in mind that even the implementation of the lower level of diffuse phosphorus source control measures may be difficult (if not impossible) in certain cases because it depends on the full cooperation of individuals and institutions.

10.3. AVAILABLE PHOSPHORUS CONTROL TECHNOLOGIES AND SUPPLEMENTARY MEASURES

During and since the monitoring phase of the OECD Programme, a large variety of phosphorus control measures have been employed in the participating projects in the Member countries. As the recovery of a lake is strongly influenced by many factors, such as morphometry, hydrology, physical, chemical and biological characteristics of water, and the nature of the drainage basin such as physical geography, edaphic conditions, climate, land use and population size, the same method of intervention is not likely to have the same result in different lakes. Furthermore the availability of financial resources would strongly influence the type of intervention that could be used. Keeping these limitations in mind, the various control methods used in the OECD Programme are briefly reviewed here. From these, water authorities can select the most suitable method(s) for their requirements.

A summary of the various sewage treatment processes, their phosphorus removal efficiency, investment and maintenance costs are given in the OECD report "Water treatment processes for phosphorus and nitrogen removal" (OECD, 1974). Since that publication, considerable development in sewage treatment has taken place which has become more economic and efficient (e.g. EPA, 1976; Baret et al. 1977).

Table 10.1

ESTIMATED COSTS OF PHOSPHORUS REDUCTION ALTERNATIVES APPLICABLE TO LAKE ERIE
(after Pluarg, 1978)

Remedial measure options	Estimated annual incremented unit costs $/kg phosphorus reduction
Urban point sources:	
Reduction of municipal sewage treatment plant effluent concentration:	
a) 1.0 mg/l to 0.5 mg/l	8.0
b) 0.5 mg/l to 0.3 mg/l	95.5
Rural nonpoint sources:	
Level 1	
Sound management on all agricultural lands, avoiding excess fertilisation, reducing soil erosion (10% phosphorus reduction)	
Level 2	
Level 1 measures, plus buffer trips, strip cropping, improved municipal drainage practices, etc., depending on region (25% reduction in phosphorus losses on soils requiring treatment).	64.3
Level 3	
Level 2 measures at greater intensity of effort (to achieve 40% reduction in phosphorus losses on soils needing treatment).	174.0
Urban nonpoint sources:	
Level 1	
Programme of pollutant reduction at source	82.0
Level 2	
Level 1 measures, plus detention/sedimentation	156.9

10.3.1. Conventional Waste Water Treatment

Conventional waste water treatment is intended to reduce the organic matter in waste water and not to control phosphorus. The purely biological and mechanical process can remove 20 - 25% of phosphorus initially present, while modified, activated sludge plants can remove about 55% of phosphorus present in some special cases. Thus, phosphorus removal efficiency of conventional waste water treatment is very limited and usually not adequate to meet the requirements of a phosphorus programme. Phosphorus removal efficiency in existing treatment plants can be improved by the application of a chemical precipitation process to the effluent.

10.3.2. Chemical Precipitation of Phosphorus

Phosphorus from waste water can be effectively eliminated with a precipitation process. In this process aluminium or iron salts or lime are added to the waste water which form insoluble compounds with the phosphates. Different kinds of precipitation processes may be employed, such as pre-precipitation, simultaneous precipitation and post-precipitation in combination with the biological process. Direct precipitation without the biological process can also be used. The precipitation process is the most effective way of reducing high quantities of phosphorus from point sources. The most comprehensive experience of phosphorus precipitation has been obtained in Sweden. In 1968 the expansion of sewage purification facilities began with the aim of removing as much as possible of the phosphorus content in waste waters (Ulmgren 1975). In early 1978 more than 600 municipal waste water treatment plans were operated with combined biological and chemical treatment.

Removal efficiencies for the different process combinations have been discussed by Grönqvist et al. (1978). It has been shown that where there is proper design and the use of suitable pH-values in the precipitation step, and no significant process disturbances, the following effluent concentrations of total phosphorus could be expected:

- pre- or simultaneous precipitations: 0.5 - 0.8 mg P/l;
- post-precipitation: 0.2 - 0.4 mg P/l;
- post-precipitation followed by filtration or simultaneous precipitation followed by contact filtration: 0.15 - 0.3 mg P/l.

However, all of the Swedish municipal waste water treatment works do not operate reliably. Investigations of post-precipitation plants indicate that effluent values of BOD; 7 are not below 15 mg/l (required limit) for about 30% of the plants and effluent values of total phosphorus are not below 0.5 mg P/l (required limit) for about 40% of the plants (National Swedish Environment Protection Board, 1977).

The unsatisfactory function of many treatment plants results partly because such plants were built too rapidly. The lack of operating experience or guidelines often resulted in the use of unsuitable process technology, building materials, equipment, etc.

However, a lot of work has been done on improving the processes and reducing the costs. The following methods, discussed by Hultman (1978), have shown promising results:

- Counterflow of precipitated sludges and two-stages precipitation;
- Regulation of the alkalinity;
- Automatic control.

After advanced waste water treatment for phosphorus removal, improved water quality has been reported for two Swedish OECD lakes, namely Lake Boren and Lake Ekoln. The phosphorus load from the treatment plants at these lakes has been reduced from 30 to 3 tons/year and from 100 to 20 tons/year, respectively, which corresponds to a decrease of the annual load of 60 - 70%. In Lake Boren and Lake Ekoln the average chlorophyll *a* values for surface water have decreased by 50%, from about 10 to 5 mg/m^3 and from 20 to 10 mg/m^3, respectively. In order to lower the lake water chlorophyll *a* concentration by an average of 1 mg/m^3 for the summer period, the annual load had to be decreased by 6 and 5 tons, respectively. In both lakes the average transparency over the summer period has increased from 1.5 to 2 m (Forsberg et al., 1978).

After reduction of phosphorus from a point source, the relative role of diffuse municipal phosphorus loading will increase, for Lake Boren for instance, from 20 to 60% of the total annual load (Forsberg et al. 1978). This means that measures against diffuse sources may be necessary if improvements cannot be achieved by precipitation of phosphorus from point sources.

10.3.3. Circular Canalisation

In certain circumstances, canalisation can be an effective way of preventing waste water entering the lakes. In well designed systems, practically all phosphorus sources can be made to bypass a lake through a circular canal. This method can only be employed where the phosphorus load which bypasses a lake will not cause problems downstream in other lakes. The advantage of this system was most effectively demonstrated in the now classic restoration case of Lake Washington (Edmondson, 1972, 1975).

10.3.4 Phosphorus Elimination in Pre-reservoirs

Pre-reservoirs are used successfully in Germany to prevent any excessive phosphorus load entering the main reservoirs used for drinking or industrial purposes (Uhlmann et al., 1971; Wilhelmus, 1976). This method is based on the principle of the sedimentation of phosphorus by algae. The elimination process is enhanced by the presence and sedimentation of inorganic particulate material upon which orthophosphate ions accumulate. The pre-reservoir should be of sufficient size and depth so that aerobic conditions are maintained and sufficient amounts of phosphorus can be sedimented. Under ideal conditions, phosphorus elimination can be as high as 70% on an average annual basis, but somewhat lower during the winter. The water retention time in the reservoir must be at least 15 days, with average flow, in order to obtain a 60% phosphorus elimination. In a chain of two or three pre-reservoirs, the phosphorus elimination can be as high as 90%.

The disadvantage of the pre-reservoir system is the fluctuation of efficiency. Phosphorus elimination can drop sharply during the winter months when algal production slows down. Conditions in long, narrow reservoirs where much of the total phosphorus is eliminated as the water moves towards the deep and, produce a similar effect to that of pre-reservoirs. The Kerr Reservoir in the United States Project is a good example.

10.3.5. Chemical Precipitation in the Lake

Iron and aluminium salts can be added directly to a lake or to its tributary inflows. This method has been applied in the reservoirs in the Netherlands (Harvelaar and Rook, 1978; Bannink and van der Vlugt, 1978); total phosphorus concentrations and algal biomass were successfully reduced in the Braakman and the Grote Rug Reservoirs. The disadvantage of this method is that some of the phosphorus precipitated is not bound permanently in the sediments and thus it could contribute to a later internal loading.

10.3.6 High Flow-through Rate

Water retention time can exert a profound influence on algal biomass. If retention time is reduced to 3 to 5 days, the algal biomass remains low regardless of the amount of nutrients available for algal growth. In certain reservoirs a short water retention time can be artificially maintained by increasing the throughout of water. Tests conducted in the Nisramont Reservoir (Shallow Lakes and Reservoirs Project) confirmed the feasibility of this approach.

10.3.7. Control of Inflow Phosphorus in the Main Tributary

Excessive phosphorus load can be removed from the main tributary of a reservoir by chemical methods. This method can be applied where the water body has no more than two tributaries. It is used in the Wahnbach Reservoir where average inflow concentration is reduced from 90 mg P/m^3 (range 50 to 200 mg P/m^3) to approximately 5 mg P/m^3 (Bernhard et al., 1971; Bernhardt, 1978). The method called the "Wahnbach System" is the combination of phosphorus precipitation with iron-III-salts, subsequent flocculation and multi-layer filtration. The system also eliminates about 77% of the chemical oxygen demand (COD), 50% of the dissolved organic carbon and 99% of the bacteria.

10.3.8 Hypolimnetic Aeration

The purpose of aeration of the tropholytic zone of thermally stratified water bodies is to artificially maintain aerobic conditions in the mud-water interface, to prevent internal loading of phosphorus and to prevent the formation of ammonium ions, sulphide ions or methane gas which are generated under anaerobic conditions. By maintaining aerobic conditions, phosphorus is retained in the sediments. Good results were obtained in the Wahnbach Reservoir with artificial aeration in the hypolimnion where dissolved oxygen concentration was maintained around 3 mg/l near the mud surface during summer stagnation (Bernhardt and Hötter, 1967; Bernhardt, 1978). Although this method does not reduce external loading, it prevents internal loading of phosphorus and enhances the elimination of phosphorus by sedimentation. It results in a lower lake concentration of total phosphorus and reduced trophic response. This method may also be used in combination with other phosphorus source control measures to speed up the apparent recovery of a lake.

10.3.9. Syphoning of Hypolumnetic Water

A syphon called an Olszewski pipe (Olszewski, 1971, 1975) is used to discharge nutrient rich water from the hypolimnion. This process reduces the thickness of the tropholytic layer and increases that of the trophogenic one, reduces the nutrient and toxic content of the hypolimnion and eliminates some of the water that is low in oxygen or lacking it completely (Ambühl, 1979). Considerable improvement in the reduction of the trophic response was obtained in several lakes such as Mauensee and Wilersee (Ambühl, 1979), and Piburgersee (Pechlaner, 1976). This method is restricted to relatively small, deep lakes with a topography suitable for the application of a syphon.

10.3.10. Artificial Water Circulation

The continuous circulation of water in sufficiently deep reservoirs with high inorganic turbidity prevents algae from remaining for long periods of time in the euphotic zone, which tends to be rather shallow because of the inorganic turbidity. By this method, low algal biomass can be maintained despite high phosphorus concentrations (e.g. Queen Elizabeth II Reservoir).

10.3.11. Reduction or Elimination of Polyphosphates from Detergent

Partial phosphorus load reduction to water bodies can also be achieved by reducing the polyphosphate content in detergent. Because of its widespread use, detergent phosphorus contributes appreciably to the municipal phosphorus load.

Apart from in a few countries (e.g. Canada and several U.S. Great Lakes border States) where low phosphorus containing detergents (2% or less) are in use, phosphate based detergents contain between 5 to 12% phosphorus by weight. Phosphorus control in detergents is actively pursued in various countries. Most OECD countries either have legislation in effect, or are considering introducting such legislation.

Because of their high biological availability, control of polyphosphates in detergents is most desirable in countries where appropriate sewage treatment facilities do not exist, and where untreated waste waters are discharged directly into important water courses and lakes. Phosphorus control in detergents is a very effective measure but should not be viewed as a substitute for the installation of appropriate waste water treatment (including tertiary treatment, i.e. phosphorus precipitation in municipal treatment plants).

10.3.12. Land Use Practices to Minimise Runoff Load, Soil Erosion and Loss of Fertiliser

Land use practices which minimise both soil erosion and extreme fluctuations in runoff water, contribute substantially to reduction of phosphorus load from diffuse sources. A considerable amount of phosphorus reduction from diffuse sources can be achieved with conscientious planning, often without extra cost. Consideration of the natural drainage in construction and agricultural practices can reduce the amount of phosphorus carried away with the runoff after rainstorms or during snow melt. These preventive measures to control diffuse sources of phosphorus can take a variety of forms. Maintaining green belts around lakes, careful storage of animal waste in feed lots, not applying fertilisers on frozen grounds, are just a few examples.

Appendix 1

REGIONAL PROJECT REPORTS

Organisation for Economic Cooperation and Development.
Cooperative Programme for Monitoring of Inland Waters (Eutrophication Control).

1. **Title : Regional Project Alpine Lakes**

   Compiled by: Hj. Fricker
   Chairmanship: M. H. Ambül, H. Löffler and O. Ravera
   Publisher: Swiss Federal Institute for Water Resources and Water Pollution Control (EAWAG) Dübendorf, Switzerland.

2. **Title : The Nordic Project : OECD Eutrophication Programme**

   Compiled by: S.-O. Ryding
   Chairmanship: C. Forsberg.
   Publisher: Nordic cooperative Organisation for Applied Research (NORDFORSK) SF-00181 Helsinki 17, Finland

3. **Title : Project, Shallow Lakes and Reservoirs. Final Report**

   Compiled by: J. Clasen
   Chairmanship: H. Bernhardt
   Publisher: Water Research Centre, Medmenham, Marlow, Bucks, SL7 2 HD, England.

4. **Title : Summary Analysis of the North American (U.S. Portion) OECD Eutrophication Project: Nutrient Loading-Lake Response Relationship and Trophic State Indices. 455 p.**

   Compiled by: W. Rast and G.F. Lee
   Project Officers: N. Jaworski and J. H. Gakstatter
   Chairmanship: N. Jaworski, M.T. Maloney and R.A. Vollenweider
   Publisher: Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Corvallis, Oregon, 97300. EPA-600/3-78-008.

5. **Title : Summary Report of the North American (Canadian Contibution) OECD Eutrophication Project**

   Prepared by: R.A. Vollenweider.

Appendix 2

THEORETICAL FRAMEWORK FOR OECD DATA ELABORATION: MASS BALANCE AND MODEL DEVELOPMENT*

1. Continuity Equation

The underlying concept developed in this report is that of a lake as an open system which can in turn be expressed as a mass balance equation, applicable to any substance involved in lake dynamics, i.e.

$$\frac{\Delta M}{\Delta t} = I - O - (S - R) \tag{1}$$

where I = external load
O = outflow loss
S = loss to sediments
R = sediment return (internal load)
and $\frac{\Delta M}{\Delta t}$ = storage gain, or storage loss over time Δt.

No hypothesis is connected to these terms which can be obtained from direct measurement. (S−R) is somewhat difficult to measure directly, but it can be obtained from the more easily measured terms in equation (1). In addition, it is possible to split these equations into two, one referring to the epilimnion, and the other to the interaction between epi- and hypolimnion (see for example Imboden 1974).

Equation (1) is the basis for any model development, although, as will be discussed later, the dimensional correctness of any derived equation is not a prerequisite as long as it is understood why dimensional correctness has been given up. In relation to some of the formulae used in this report, it is important to keep this preamble in mind in relation to some of the model terms which for practical reasons, have been derived statistically.

2. Steady State

The steady state equation is a particular form of equation (1) by setting $\frac{\Delta M}{\Delta t} = 0$. In reality no lake is in a steady state over short or even long periods of time. However, lakes, the nutrient load of which oscillates around a constant mean value over prolonged periods (several years), can be thought to exist in a repetitive steady definied by e.g. the storage content measured at spring overturn time, or by the yearly average storage content. Both possibilities have been explored in specific data elaboration.

3. Model Development

3.1. Background

In the model development pursued in this report, the basic hypothesis introduced was that a lake is considered as a completely *mixed reactor* (CMR). This

(*) Prepared by R.A. Vollenweider.

assumption is only partially true for most lakes. In some cases, however, it would be more appropriate to treat a given lake either as a plug flow reactor (PFR), or as a sequence of coupled mixed reactors.

None of these models is entirely satisfactory for the variety of situations considered in this report. On the other hand the multistage reactor concept, an important basis for dynamic modelling, is difficult to apply for comparative purposes as envisaged in the OECD Programme. Therefore, a deliberate choice has been made, restricting the approach to the mixed reactor concept, combining its application in data elaboration with statistical techniques. Accordingly, the resulting relationships are a means by which to describe the average statistical behaviour of a large population of lakes, rather than the specific behaviour of a single lake. On the other hand, from the knowledge of the statistical behaviour of a large population of lakes, it is possible to obtain a measure of the degree of deviation of a particular lake from the total population which permits rapid identification of specific situations.

3.2. Instantaneous Mass Balance Equation

At any one time, the instantaneous change of mass storage in a lake can be derived from the general mass balance equation (1), i.e.

$$\frac{d M_{\lambda,t}}{dt} = \Sigma \left(Q_{i,t}[M]_{i,t} - Q_{\omega,t}[M]_{\omega,t} - A_e F_e(M) + A_s F_s(M)\right) \qquad (2)$$

in which:

$Q_{i,t}$ = volume of the i-th inflow at time t (m^3/sec)
$[M]_{i,t}$ = concentration of the i-th inflow at time t (mg/m^3)
$Q_{\omega,t}$ = outflow volume at time t (m^3/sec)
$[M]_{\omega,t}$ = concentration of the outflow at time i (mg/m^3)
A_e and A_s = the areas of the epilimnion and the sediments, respectively
$F_e(M)$ and $F_s(M)$ = the positive and negative fluxes through these areas of substance M ($mg/m^2.sec$).

The solution of this equation presupposes knowledge of each term through appropriate measurement.

In order to solve this equation, a few assumptions and simplifications are necessary. Instead of considering the instantaneous processes, equation (2) is better written as a difference equation over time Δt, where Δt, for our purpose has been assumed to be one year. In this case, ΣQ_i becomes the total water discharged over this period and in order to obtain the mass involved, the concentration may be approximated with a term denoting an "average concentration".

The same procedure is further applied to the other terms. In particular, $A_e F_e(M)$ may be approximated, introducing the hypothesis $A_e F_e(M) = A_e \bar{s}\,[M]_\lambda$, where $[M]_\lambda$ is an average lake concentration, and s an apparent average settling rate (m/y). This latter is substantially different from instantaneous settling rates.

3.3. General Calculation Model

With further simplification, and defining

$$[\overline{M}]_\omega \equiv [\overline{M}]_\lambda, \qquad \frac{\Sigma\Sigma Q_i[\overline{M}]_i}{\Sigma\Sigma Q_i} \equiv [\overline{M}]_i \quad \text{et} \quad \Sigma Q_i \equiv \Sigma Q_\omega \equiv \overline{Q}_y$$

equation (2) can be written

$$\frac{\Delta M}{\Delta t} = \overline{Q}_y[\overline{M}]_j - \overline{Q}_y[\overline{M}]_\lambda - A_e \bar{s} M_\lambda + A_s F_s(M) \qquad (3)$$

or, by dividing both sides by V_0, the lake volume,

$$\frac{I\Delta M}{V_0 \Delta t} = \left(\frac{Q_y}{V_0}\right)[\overline{M}]_j - \left(\frac{\overline{Q}_y}{V_0}\right)[\overline{M}]_\lambda - \left(\frac{A_e}{V_0}\right)\overline{s}[\overline{M}]_\lambda + \left(\frac{A_s}{V_0}\right)F_s(\overline{M}) \tag{3a}$$

For steady state conditions (always defining steady state in the sense defined in (§ 2), the resulting average lake concentration is:

$$[M]_\lambda \simeq \frac{\left(\frac{Q_y}{V_0}\right)[M]_j + \left(\frac{A_s}{V_0}\right)Fs(M)}{\left(\frac{Q_y}{V_0}\right) + \left(\frac{A_e}{V_0}\right)\overline{s}} \tag{4}$$

The enumerator in (4) represents the total volumnar load (external and internal) of the lake with substance M, and the denominator represents an elimination function.

3.4. Specific Calculation Models

In proceeding to considerations of specific models, it should be noted first, by definition $(V_0/\overline{Q}_y) = T(w)$, $\tau(w)$ is synonymous with the filling time related to the yearly outflow volume. Furthermore A_e and A_s, and (V_0/A_e) and (V_0/A_s) are close to surface area A_0. and the mean depth z of the lake, respectively, yet not entirely identical. This distinction is important for understanding the variations in specific calculation formulae produced from the OECD data. Additional *definition* terms can be mentioned:

$\frac{\overline{Q}_y}{A_0} = q_a$	=	hydraulic load (m/y)
$[\overline{M}]_\lambda \cdot q_a = L(M)$	=	areal loading to the lake of substance M ($mg/m^2.y$)
$\frac{L(M)}{\overline{z}} = \ell(M)$	=	the volumnar loading to the lake of substance M ($mg/m^3.y$) which latter is identical to Edmondson's potential loading (Edmondson 1961).
$T(w) = \frac{V_0}{Q_y} = \overline{z}/q_a$	=	reciprocal of the flushing rate/year, σ.

Accordingly, these definition terms are interchangeable, and equation (4) may assume different forms but have identical content. For example, by multiplying numerator and denominator of (4) by $(V_0/\overline{Q}_y)$, and assuming that $A_e \approx A_s \approx A_0$, it follows that:

$$[\overline{M}]_\lambda = \frac{[\overline{M}]_j + \frac{T(w)}{z}.Fs(M)}{1 + \frac{T(w)}{\overline{z}}} \equiv \frac{[\overline{M}]_j + \frac{Fs(\overline{M})}{q_a}}{1 + \frac{\overline{s}}{q_a}} \equiv \frac{[\overline{M}]_j q_a + Fs(M)}{q_a + \overline{s}}$$

However, the above assumption is somewhat arbitrary. In order to avoid inappropriate rigidity, it is better to generalise (4) to

$$[\overline{M}]_\lambda = \frac{[\overline{M}]_j + Fs(M).f_1(T(w),z)}{1 + \overline{s}.f_2(T(w),\overline{z})} = \frac{[\overline{M}]_j + Fs(M).f_1(q_a,\overline{z})}{1 + \overline{s}.f_2'(q_a,\overline{z})} \tag{4a}$$

A particularly important limiting case of (4a) is given for $f_1 = 0$, i.e. internal loading is considered zero. Then

$$[\overline{M}]_\lambda = \frac{[\overline{M}]_j}{(1 + \overline{s}\,f(T(w),\overline{z})} \tag{4b}$$

$\overline{s}.f(T(w),z)$ can then be obtained by statistical analysis using the equivalency

$$\frac{[M]_\lambda}{[M]_j} = \pi_r = \frac{1}{1 + \overline{s}f(T(w),\overline{z})}$$

This usually leads to non-linear terms in T(w) and z.

Vollenweider (1969, 1975, 1976) and Vollenweider and Dillon (1974), Dillon and Rigler (1974) have explored equation (4b) for phosphorus. The original equation of Vollenweider (1969)

$$[\overline{P}]_\lambda = \frac{C(P)}{\overline{z}\ \frac{1}{T(w)} + \sigma}$$

can indeed be expressed in the form of (4b) as

$$[\overline{P}]_\lambda = [\overline{P}]_j/\big(1 + \sigma\, T(w)\big) \tag{5}$$

and the same form applies also to his more recent modification

$$[\overline{P}]_\lambda = [\overline{P}]_j/\big(1 + \sqrt{T(w)}\big)] \tag{6}$$

While in equation (5) the dimensional meaning of σ was equivalent to $\overline{s}/\overline{z}$, $\overline{s}$ being 10 to 20 m/y, equation (6) would be equivalent to (5) if $\sigma = \sqrt{T(w)}$'. Both relationships have been derived from statistical elaboration of data. The initial success of equation (6) made it a reference equation widely used in this report. Necessary modifications will be discussed in appropriate places. However, it also has been shown, particularly from the reservoir data, what equation (4b) is more appropriate for many cases.

Another version of equation (4b) has been considered by Larsen and Mercier (1976) and Dillon and Rigler (1974). Considering that the retention coefficient is defined as

$$R = 1 - [\overline{P}]_o/[\overline{P}]_j\ , \tag{7}$$

and assuming also that $[\overline{P}]_o = [\overline{P}]_\lambda$, the lake concentration may also be derived from

$$[\overline{P}]_\lambda = [\overline{P}]_j\ (1-R) \tag{6a}$$

From approximations made similar to those pursued by Vollenweider, Larson and Mercier concluded that the best statistical fit to data resulted from setting

$$R = \frac{1}{1 + \sqrt{(1/T(w)}} \tag{7a}$$

Introducing (7a) into (6a), this latter becomes equivalent to (6).

Dillon and Rigler in their version, on the other hand, do not attempt to define R independently, but using R_{exp} to calculate σ appearing in equation (5),

$$\sigma = \frac{R\rho}{1-R}\ ,$$

ρ being 1/T(w).

From which follows

$$[P]_\lambda = \frac{L(P)\ (1-R)}{\overline{z}\rho} \tag{8}$$

However, it is to be noted that in this form, (8) does not predict the lake concentration from load, but is rather a test of the hypothesis that outflow concentration equals lake concentration. This derives from the definition of R as given in 7. Therefore, (8) cannot be used for predictive purposes. The problem is solved, however, with Kirchner and Dillon's (1975) statistical elaboration of R from q_a ,

$$R = 0.426\ \exp(-0.271\ q_a) + 0.574\ \exp(-0.00949\ q_a) \tag{9}$$

which is an independent estimate of R.

Modifications of (4b) have been considered further by Chapra (1975) and Reckhow (1979). Reckhow introduced uncertainty analyses with which the statistical validity of model terms are evaluated. In addition, he attempted to separate lakes into three classes giving specific situations for each class, besides a quasi-general solution. Reckhow's formulae have not been tested on OECD data, for details the reader should consult the original publication. However, it should be noted that Reckhow's attempt at further development of empirical solutions for (4b) is of considerable importance.

3.5. Model Development including Biological Parameters

The relationship between lake concentration and biological parameters, such as chlorophyll, is not a question of models, but of straightforward statistical analysis. The situation is different, however, when considering the relationship between loading and biological response, yet, the principles are the same as used for predicting lake concentration from load.

Thomann (1977, 1978) and Schindler (1978) have raised this point. Thomann showed that empirical solutions for equations corresponding to (4b) are special cases of more complex system models, and that their applicability for predictive purposes depends on whether one can quantify reliably the relationship mentioned above, i.e if the ratio [M-Phytoplankton]/[M total] remains more or less constant.

The OECD Programme has shown that this is indeed the case within certain limits for the chlorophyll/phosphorus relationship.

For model development, therefore, it follows that (4b) must be a reasonable predictor for chlorophyll, if the appropriate ratio is introduced, and phosphorus is the controlling factor, i.e.

$$[\mathrm{Chl}] \approx \frac{\alpha[\mathrm{P}]_j}{\left(1 + \overline{s}.\ f(T(w),\overline{z})\right)} \qquad (10)$$

Vollenweider (1976), and Rast and Lee (1978) have shown that this relationship holds within certain boundaries using standard predictor $1/\left(1 + \sqrt{T(w)}\right)$. Necessary modifications which have arisen from the OECD results are considered in appropriate sections of the report.

Besides chlorophyll, modifications of (10) are applicable to primary production.

4. Loading Criteria

Loading criteria may be derived from rewriting (4a) and (10). Their primary aim is to relate loading to qualitative lake characteristics as defined by the trophic categories. However, it is now also possible to ignore their trophic connotation, and to use model relationships only in the sense of a gliding scale relationship. This permits presenting of any arbitrary goals, and evaluation from these of the kind of loading which has to be met to achieve such goals.

Vollenweider (1968) made the first attempt to formulate loading criteria for phosphorus (and to some extent for nitrogen) in the first sense given above, by defining the separation between oligotrophic and eutrophic lakes with regard to loading relative to mean depth, according to

$$L(P)_c\ (\mathrm{mg\ P/m^2.y})\ (25\ \text{to}\ 50)\ \overline{z}^{0.6}$$

This restriction to mean depth invalidates applicability to lakes of an essentially complex hydrological regime. To make loading criteria more general, it is necessary to include, besides mean depth, water residence time (or conversely, the hydrological load) as an essential parameter loading criteria, i.e.

$$L(P)_c = f(T(w),\overline{z})$$

In line with this, it is logical to postulate that the critical Loading L_c (which can also be interpreted as loading tolerance) for a given lake is directly proportional to a function of the residence time (response directly proportional to a function of the flushing rate).

To account for this, Vollenweider (1975, 1976), proposed two versions.

$$L_c\ (\text{mg P/m}^2.\text{y}) \approx \overline{[P]}_{\lambda,c} \left(\frac{\bar{z}}{T(w)}\right) (1 + K\tau_w)$$

and

$$L_c\ (\text{mg P/m}^2.\text{y}) \approx \overline{[P]}_{\lambda,c} \left(\frac{\bar{z}}{T(w)}\right) (1 + \sqrt{\tau_w})$$

$[P]_{\lambda,c}$ being a critical concentration between 10 and 20 mg P/m^3 for separating oligotrophic from eutrophic waters. Obviously, the second version is more specific and requires no assumptions in the choice fo value for K.

Both equations are consistent with (4b) which, for the purpose of defining loading criteria, can be written either as

$$[P]_{j.c} \approx [P]_{\lambda,c} [1 + \bar{s}\, f\, (T(w), z)] \tag{11}$$

or

$$L(P)_c \approx [P]_{\lambda,c} \left(\frac{\bar{z}}{T(w)}\right) [1 + \bar{s}\, f\, (T(w), \bar{z})] \tag{11a}$$

Corresponding considerations, derived from two-layer models, have also been made by Imboden (1974) and Snodgrass and O'Melia (1975).

Finally, it is apparent that instead of prefixing a critical concentration level for phosphorus (i.e. $[P]_{\lambda,c}$), the same relationship can be used to derive phosphorus loading criteria as they relate to "desirable" biomass (chlorophyll) level. Although conceptually identical with the former, it leads immediately to a more flexible consideration of loading criteria, since the objective set is not a prefixed nutrient concentration but a biomass level compatible with water use objectives. For different water uses such objectives may differ considerably. For example, criteria relating to use of a reservoir for drinking water supply are likely to be more stringent than criteria for only recreational use of a water body.

5. Time Response

In relation to time response, one has to consider that —with changing conditions of loading - the actual lake concentration is not in equilibrium with the actual loading. After the loading has stabilised over a prolonged period, the lake attains a new dynamic equilibrium, yet the velocity at which this equilibrium state is reached is somewhat difficult to estimate.

5.1. An estimate for the time response can be obtained from the assumption that variation with time follows the equation

$$\frac{d[M]'_\lambda}{[M]_{eqv} - [M]'_\lambda} = \frac{1}{\overline{T(M)}} \cdot dt \tag{12}$$

in which

- $[M]'_\lambda$ = the actual lake concentration;
- $[M]_{eqv}$ = theoretical equilibrium concentration;
- $\overline{T(M)}$ = bulk residence time of substance M accounting for all losses and internal sources.

This latter represents the lumping into one term of the last three terms of equation (2). The equilibrium concentration $[M]_{eqv}$ is defined as

$$[M]_{eqv} = \frac{\overline{T}(\overline{M})}{\overline{T}(w)} \cdot [M]_j \tag{12a}$$

T(M)/T(w) represents a relative residence time, i.e. the average bulk residence time of substance M relative to the average residence time of water.

The time variation in (12), therefore, is positive when $[M]_{eqv} > [M]'_\lambda$, and *vice versa*, becomes negative when $[M_{eqv} > [M]'_\lambda$.

5.2. Integration of (12) gives

$$[M]_{\lambda}^{\prime\prime} = [M]_{eqv} + \left((M)_{\lambda}^{\prime} - (M)_{eqv}\right) . e^{-}\left(\frac{t}{\overline{T(M)}}\right) \tag{13}$$

For the time resolution of $t = 1$, (13) predicts the concentration $[M]_{\lambda}^{\prime\prime}$ for the following year. Accordingly, this equation can be solved stepwise for any new loading condition which, however, requires re-calculation for each step of $[M]_{eqv}$ according to (12).

5.3. A crude estimate of the response time for the lake concentration to attain 95% of the equilibrium concentration can further be obtained from setting

$$t(M)_{95} = \frac{3}{\frac{1}{T(w)} + \sigma} = 3 \frac{V_0}{\overline{Q} + \sigma V_0} \tag{14}$$

(see Vollenweider 1969, Chapra and Dobson in press). Here σ has the meaning given in 3.4. Using the better defined experimental retention coefficient instead of σ, (14) equals also

$$t(M)_{95} = 3 . T(w) . (1 - R_{exp}) \tag{14a}$$

R_{exp}, of course, can also be substituted by R of equation (9).

Appendix 3

PROCEDURES FOR DATA SCREENING

1. General

The treatment of the reported data varies somewhat between the synthesis report and the regional projects. The regional project reports treat each year of observation as a separate data point, thus, a given water body may be represented by more than one point in the analysis. This means that if a lake has only one year of monitoring reported, it is represented by one point in the analysis while a lake with three years of observations is represented by three. Thus, a lake with several years of data carries more weight in the regression analysis. In the synthesis report the data was averaged when two or more years of data were available, with the exception of Lake Minetonka and Lake Washington, in the USA Project. These lakes underwent nutrient load reduction and the years of observation represent before and after nutrient reduction conditions. Therefore, Lake Minetonka is represented by two data points (1969 and 1975) and Lake Washington by three (1957, 1964, and 1971).

There are certain advantages in using mean values for more than one year of data: over-representation of water bodies with several years of data can be reduced; and by using mean values and one data point for the analysis, erratic lake responses caused by changing nutrient load or hydraulic load are dampened. The figure thus obtained approximates more closely an equilibrium, steady state condition which is more appropriate for use in the nutrient load formulae employed in the data analysis. The effects of changing nutrient load (for example Lake Esrøm, Nordic Project) can be seen in the regional reports where the data was treated separately for each year of observation.

The Shallow Lakes and Reservoirs Project incorporated the reservoirs from the USA Project in the data analysis to obtain a broader data base. Thus, the USA reservoirs are considered in two regional projects.

2. Correlations

2.1. Unscreened data

Annual mean values for all available data points were used. If the annual mean value was not available, the lake in question was automatically omitted from analysis for that variable.

2.2. Screened data

Screening procedures given in § 3.2 were applied.

3. Regression Analysis

3.1. Unscreened data

Same as 2.1.

3.2. Screened data

3.2.1. Chlorophyll a concentration

a) Three lakes, where the algal biomass was suppressed artificially or by natural turbidity, were excluded from all chlorophyll relationships (nos 331, 312, 328, in Table 2.2.).

b) In the final run of chlorophyll relationships lakes with inorganic-N/ortho-P ratios < 10 were excluded (101, 205, 206, 207, 208, 209, 210, 211, 212, 216, 228, 229, 428, 430, 447, 450).

3.2.2. Phosphorus loading

a) Lakes with $[\overline{P}]_\lambda > [\overline{P}]_j$, where internal loading was assumed or was known to exist, were excluded from further analysis (213, 214, 218, 221, 226, 302, 428, 429, 441, 443).

b) Lakes with high T(w) ($>$ 10) with $[\overline{P}]_\lambda \approx [\overline{P}]_j$, where internal loading was assumed were excluded from further analysis (e.g. Lac Léman 228).

c) Estuaries and long narrow reservoirs where the average lake concentration given was not representative of the true, average inlake concentration were excluded (416, 417, 428, 429).

d) Lake Annone (232) was excluded because $[\overline{P}]_\lambda$ was greatly reduced as a result of industrial pollution.

e) Grafham Water (319) was excluded because the inlake concentration of total phosphorus predicted from loading was more than four times greater than the inlake concenctration measured at spring overturn.

3.2.3. Nitrogen loading

Lakes excluded from 3.2.2. (phosphorus loading) were excluded.

3.2.4. Primary Production

a) Excluded from *all* relationships were lakes where the annual primary production was > 800 g C/m^2.y and lakes 114, 201 and 213.

b) No 328 was excluded from the relationship. "Primary production versus annual mean total phosphorus and flushing corrected annual inflow phosphorus concentrations".

c) No 332 was excluded from "primary production versus annual mean chlorophyll".

4. Assessment of trophic state

The trophic statue was assessed from a separate data base provided by the regional consultants and extracted from the USA Project. All available data points were used. In cases where an annual mean value was not available, it was estimated from the seasonal value(s) where appropriate.

Note: In the synthesis report in addition to the regressions calculated for the combined OECD data (Tables 6.1 and 6.2), in most cases regressions were recalculated for each regional project using the screening procedure given in § 3 of this Appendix (Table 6.2). Therefore, for each regression, if available, two equations are given for each regional project. The first regression is obtained from the data base used for this report (see § 1 of this Appendix) and the second regression is from the appropriate regional report (Appendix 1).

Appendix 4

STATISTICAL FEATURES OF THE DATA USED FOR CORRELATIONS AND REGRESSION ANALYSES

In the following Table, for each relationship a reference is given to the appropriate Chapter, Figure or Table, along with the symbol of the two variables x and y. The following statistics are given for each:

n	=	number of data points
r	=	correlation coefficient
$\bar{x}$, $\bar{y}$	=	the mean of variable x and y
s_x^2 , s_y^2	=	variance of x and y ; and range of x and y.

Table A.4.1.

Référence	Variables		n	r	Log 10					
	y	x			$\overline{x}$	s_x^2	Valeurs extrêmes de x	$\overline{y}$	s_y^2	Valeurs extrêmes de y
Figure 4.1	$\overline{[N]}_\lambda$	$\overline{[P]}_\lambda$	57	0.75	1.689	0.294	0.637 - 2.587	3.089	0.147	2.421 - 3.785
Figure 4.2	$\overline{[PO_4\text{-}P]}_\lambda$	$\overline{[P]}_\lambda$	92	0.89	1.677	0.266	0.637 - 2.875	1.169	0.522	- 0.699 - 2.699
Figure 4.3	$\overline{[In\text{-}N]}_\lambda$	$\overline{[N]}_\lambda$	38	0.95	3.101	0.127	2.486 - 3.785	2.930	0.159	2.241 - 3.785
Figure 4.4	$\overline{[In\text{-}N]}_\lambda$	$\overline{[PO_4\text{-}P]}_\lambda$	89	0.47	1.253	0.571	- 0.699 - 2.950	2.760	0.230	1.699 - 3.732
Table 4.1	$\overline{[chl]}$	$\overline{[P]}_\lambda$	99	0.75	1.653	0.290	0.477 - 2.875	0.924	0.248	- 0.523 - 1.954
Table 4.1	$\overline{[chl]}$	$\overline{[P]}_\lambda$	88	0.88	1.579	0.275	0.477 - 2.875	0.942	0.275	- 0.523 - 1.954
Table 4.2	$\overline{[chl]}$	$\overline{[P]}_\lambda$	77	0.88	1.490	0.232	0.477 - 2.462	0.890	0.277	- 0.523 - 1.889
Table 4.1	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[P]}_\lambda$	65	0.70	1.651	0.302	0.637 - 2.587	1.348	0.270	0.301 - 2.440
Table 4.1	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[P]}_\lambda$	54	0.86	1.529	0.273	0.637 - 2.587	1.364	0.316	0.301 - 2.440
Table 4.2	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[P]}_\lambda$	50	0.90	1.468	0.241	0.637 - 2.462	1.342	0.327	0.301 - 2.440
Table 4.1	$\overline{[chl]}$	$\overline{[N]}_\lambda$	53	0.47	3.106	0.148	2.421 - 3.785	0.863	0.201	- 0.155 - 1.889
Table 4.1	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[N]}_\lambda$	52	0.50	3.104	0.151	2.421 - 3.785	1.341	0.243	0.301 - 2.440
Table 4.2	$\overline{[chl]}$	$\overline{[N]}_\lambda$	41	0.64	2.986	0.123	2.421 - 3.785	0.882	0.255	- 0.155 - 1.889
Table 4.2	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[N]}_\lambda$	40	0.66	2.982	0.125	2.421 - 3.785	1.355	0.302	0.301 - 2.440
§ 4.5	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[chl]}$	73	0.93	0.864	0.220	- 0.155 - 1.889	1.347	0.271	0.301 - 2.440
Figure 6.3	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]$	$\overline{[chl]}$	72	0.95	0.873	0.218	- 0.155 - 1.889	1.343	0.273	0.301 - 2.444
Figure 4.6	$\overline{[chl]}/\overline{[P]}_\lambda$	$\overline{[P]}_\lambda$	77	- 0.04	1.490	0.232	0.477 - 2.462	0.293	0.026	0.080 - 0.710
Figure 4.7	$\left[\begin{matrix}\text{max}\\ \text{chl}\end{matrix}\right]/\overline{[P]}_\lambda$	$\overline{[P]}_\lambda$	50	0.11	1.468	0.241	0.637 - 2.462	0.888	0.300	0.250 - 2.520
§ 4.7	$\overline{[Sec]}$	$\overline{[chl]}$	78	- 0.75	0.944	0.192	- 0.523 - 1.880	0.490	0.088	- 0.097 - 1.452
§ 4.7	$\overline{[Sec]}$	$\overline{[P]}_\lambda$	87	- 0.47	1.708	0.261	0.477 - 2.735	0.511	0.095	- 0.097 - 1.452
§ 4.8	ΣPP	$\overline{[chl]}$	57	0.74	0.921	0.241	- 0.523 - 1.889	2.304	0.219	0.748 - 3.067
§ 4.8	ΣPP	$\overline{[P]}_\lambda$	57	0.71	1.606	0.254	0.477 - 2.735	2.333	0.212	0.748 - 3.041
Figure 5.1	$\overline{[P]}_\lambda$	$\overline{[P]}_j$	101	0.82	2.062	0.308	0.670 - 3.154	1.626	0.295	0.478 - 2.875
§ 5.3 combined (screened)	$\overline{[P]}_\lambda$	$\overline{[P]}_j$	87	0.87	2.062	0.300	0.670 - 3.154	1.591	0.273	0.477 - 2.544
Nordic	$\overline{[P]}_\lambda$	$\overline{[P]}_j$	14	0.73	1.936	0.233	1.359 - 2.799	1.453	0.211	0.903 - 2.377
Alpine	$\overline{[P]}_\lambda$	$\overline{[P]}_j$	18	0.92	1.958	0.367	0.670 - 2.858	1.506	0.281	0.637 - 2.316
S L & R	$\overline{[P]}_\lambda$	$\overline{[P]}_j$	24	0.91	2.163	0.319	1.007 - 3.022	1.710	0.293	0.669 - 2.519

Référence	Variables		n	r	Log 10					
	y	x	n	r	$\overline{x}$	s_x^2	Valeurs extrêmes de x	$\overline{y}$	s_y^2	Valeurs extrêmes de y
USA	$[\overline{P}]_d$	$[\overline{P}]_j$	31	0.84	2.100	0.262	0.997 - 3.154	1.601	0.265	0.477 - 2.544
combined (screened)	$[\overline{P}]_\lambda$	$\frac{[\overline{P}]_j}{1+\sqrt{T(w)'}}$	87	0.93	1.700	0.350	0.250 - 2.822	1.591	0.273	0.477 - 2.544
Nordic	»	»	14	0.86	1.530	0.186	0.940 - 2.469	1.454	0.211	0.903 - 2.377
Alpine	»	»	18	0.93	1.579	0.351	0.439 - 2.656	1.506	0.281	0.637 - 2.316
S L & R	»	»	24	0.95	1.917	0.332	0.781 - 2.822	1.710	0.293	0.669 - 2.519
USA	»	»	31	0.95	1.670	0.390	0.250 - 2.781	1.601	0.265	0.477 - 2.544
Combined	$[\overline{N}]_\lambda$	$\frac{[\overline{N}]_j}{1+\sqrt{T(w)}}$	46	0.92	2.968	0.204	2.148 - 3.867	3.033	0.142	2.421 - 3.785
Combined (screened)	»	»	42	0.92	2.982	0.205	2.148 - 3.867	3.045	0.148	2.421 - 3.785
Table 5.1	$[\overline{chl}]$	»	91	0.78	1.700	0.379	0.250 - 3.158	0.911	0.259	- 0.523 - 1.954
	$\begin{bmatrix}max\\chl\end{bmatrix}$	»	58	0.77	1.704	0.388	0.439 - 3.158	1.361	0.286	0.301 - 2.440
	$[\overline{chl}]$	$\frac{[\overline{N}]_j}{1+\sqrt{T(w)'}}$	48	0.60	2.944	0.218	1.849 - 3.867	0.895	0.246	- 0.145 - 1.880
	$\begin{bmatrix}max\\chl\end{bmatrix}$	»	47	0.65	2.936	0.220	1.844 - 3.867	1.385	0.299	0.301 - 2.440
Table 5.2	$[\overline{chl}]$	$\frac{[\overline{P}]_j}{1+\sqrt{T(w)'}}$	40	0.86	1.678	0.334	0.447 - 2.822	0.900	0.281	- 0.155 - 1.880
	$\begin{bmatrix}max\\chl\end{bmatrix}$	»	39	0.88	1.671	0.341	0.447 - 2.822	1.360	0.348	0.301 - 2.439
	$[\overline{chl}]$	$\frac{[\overline{N}]_j}{1+\sqrt{T(w)'}}$	40	0.68	2.892	0.220	1. 844 - 3.867	0.900	0.281	- 0.155 - 1.880
	$\begin{bmatrix}max\\chl\end{bmatrix}$	»	39	0.69	2.881	0.221	1.844 - 3.867	1.361	0.348	0.301 - 2.440
§ 5.5	ΣPP	$\frac{[\overline{P}]_j}{1+\sqrt{T(w)'}}$	42	0.72	1.617	0.252	0.610 - 2.656	2.295	0.272	0.748 - 3.067
Table 6.1										
A1	$[\overline{chl}]$	$[\overline{P}]_\lambda$	99	0.75	See Table 4.1					
A2	»	»	88	0.85	See Table 4.1					
A3	»	»	77	0.88	See Table 4.2 (fig. 6.1)					
B1	$\begin{bmatrix}max\\chl\end{bmatrix}$	$[\overline{P}]_\lambda$	65	0.70	See Table 4.1					
B2	»	»	54	0.86	See Table 4.1					
B3	»	»	50	0.90	See Table 4.2 (fig. 6.2)					
C1	»	$[\overline{chl}]$	73	0.93	See § 4.5					
C2	$\begin{bmatrix}max\\chl\end{bmatrix}$	$[\overline{chl}]$	72	0.95	See § 4.5 (fig. 6.3)					
Table 6.2										
Combined	$[\overline{chl}]$	$[\overline{P}]$	77	0.88	See Table 4.2 (fig. 6.1)					
Nordic	»	»	14	0.93	1.442	0.191	0.903 - 2.213	0.732	0.357	- 0.155 - 1.889
Alpine	»	»	16	0.83	1.369	0.199	0.637 - 2.155	0.880	0.221	0.123 - 1.880
S L & R	»	»	21	0.90	1.579	0.280	0.669 - 2.462	1.002	0.228	0.021 - 1.699
USA	»	»	26	0.91	1.518	0.216	0.477 - 2.398	0,891	0.284	- 0.523 - 1.530
Combined	$\begin{bmatrix}max\\chl\end{bmatrix}$	$[\overline{P}]_\lambda$	50	0.90	See Table 4.2 (fig. 6.2)					
Nordic	»	»	14	0.94	1.442	0.191	0.903 - 2.213	1.252	0.363	0.301 - 2.349
Alpine	»	»	16	0.90	1.369	0.199	0.637 - 2.155	1.366	0.285	0.380 - 2.347
S L & R	»	»	20	0.91	1.566	0.291	0.669 - 2.462	1.385	0.328	0.312 - 2.440
Figure 6.4	$[\overline{Sec}]^y$	$[\overline{chl}]$	78	- 0.75	See § 4.7					
Figure 6.5	$[\overline{Sec}]^y$	$[\overline{P}]_\lambda$	87	- 0.45	See § 4.7					

Référence	Variables		n	r	Log 10					
	y	x	n	r	$\overline{x}$	s^2_x	Valeurs extrêmes de x	$\overline{y}$	s^2_y	Valeurs extrêmes de y
Figure 6.7	ΣPP	»	49	0.71	1.612	0.239	0.477 - 2.735	2.364	0.137	0.748 - 2.884
Figure 6.9	»	$[\overline{chl}]$	49	0.79	0.955	0.247	0.523 - 1.889	2.338	0.149	0.748 - 2.887
Table 6.3	$[\bar{P}]_\lambda$	$[\bar{P}]_j$			See § 5.3					
Table 6.4	$[\bar{P}]_\lambda$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	101	0.82	1.704	0.356	0.250 - 2.822	1.626	0.295	0.477 - 2.875
»	»	»	87	0.87		See § 5.3 (fig. 6.11)				
»	$[\bar{N}]_\lambda$	$\frac{[\bar{N}]_j}{1+\sqrt{T(w)'}}$	46	0.92	2.968	0.204	2.148 - 3.867	3.033	0.143	2.421 - 3.785
»	»	»	42	0.92		See § 5.3 (fig. 6.12)				
Table 6.5	$[\bar{P}]_\lambda$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$				See § 5.3 (fig. 6.11)				
Figure 6.13	$[P]_o$	$[P]_\lambda$	62	0.95	1.604	0.293	0.637 - 2.735	1.607	0.276	0.574 - 2.744
Figure 6.14	$[\bar{P}$	$[\bar{P}]_e$	59	0.96	1.549	0.265	0.580 - 2.512	1.590	0.276	0.574 - 2.744
Table 6.6 .										
A1	$[\overline{chl}]$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	91	0.78		See § 5.4 (Fig. 6.15)				
A2	»	»	74	0.87	1.638	0.345	0.250 - 2.822	0.870	0.271	- 0.522 - 1.880
A3	»	»	67	0.88	1.589	0.344	0.250 - 2.822	0.835	0.281	- 0.522 - 1.880
B1	$\begin{bmatrix} max \\ chl \end{bmatrix}$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	58	0.77		See § 5.4 (Fig. 6.16)				
B2	»	»	47	0.88	1.656	0.328	0.439 - 2.822	1.341	0.336	0.301 - 2.440
B3	»	»	45	0.88	1.631	0.328	0.439 - 2.822	1.325	0.339	0.301 - 2.440
Table 6.7										
Nordic	$[\overline{chl}]$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	13	0.82	1.485	0.172	0.940 - 2.469	0.643	0.273	- 0.155 - 1.710
Alpine	»	»	12	0.94	1.470	0.395	0.439 - 2.656	0.809	0.268	0.124 - 1.880
S L & R	»	»	20	0.89	1.504	0.385	0.250 - 2.400	0.770	0.299	- 0.523 - 1.530
Nordic	$\begin{bmatrix} max \\ chl \end{bmatrix}$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	13	0.77	1.485	0.172	0.940 - 2.469	1.168	0.291	0.301 - 2.270
Alpine	»	»	11	0.96	1.505	0.416	0.439 - 2.656	1.306	0.382	0.380 - 2.340
S L & R	»	»	21	0.88	1.787	0.333	0.781 - 2.822	1.432	0.318	0.312 - 2.440
§ 6.42	$[\overline{chl}]$	$\frac{[\bar{N}]_j}{1X\sqrt{T(w)'}}$			See Table 5.2					
	$\begin{bmatrix} max \\ chl \end{bmatrix}$	»			See Table 5.2					
Table 6.8 (Fig. 6.17)										
Combined	$[\overline{Sec}]^y$	$\frac{[\bar{P}]_j}{1+\sqrt{T(w)'}}$	67	- 0.69	1.797	0.325	0.439 - 2.822	0.474	0.102	- 0.097 - 1.452
Alpine	»	»	18	- 0.74	1.579	0.352	0.439 - 2.656	0.713	0.058	0.121 - 0.954
S L & R	»	»	26	- 0.55	1.887	0.326	0.781 - 2.822	0.429	0.075	0.067 - 1.067
USA	»	»	22	- 0.82	1.860	0.264	0.610 - 2.781	0.348	0.105	- 0.097 - 1.457
§ 6.44	ΣPP	»	34	0.75	1.587	0.258	0.610 - 2.656	2.310	0.166	0.748 - 2.884

Appendix 5

SELECTED CALCULATIONS USING THE STATISTICAL FEATURES IN APPENDIX 4

1. Statistical features given in Appendix 4:

$\bar{x}, \bar{y}$ = ranges of x and y
n = number of data points
s_x^2 = x variance
s_y^2 = y variance
SE = standard error of estimate
r = correlation coefficient

2. Statistical formulae for the features listed above:

$$\bar{x} = \frac{\Sigma x_i}{n} \quad ; \quad \bar{y} = \frac{\Sigma y_i}{n}$$

$$s_x^2 = \frac{\Sigma(x_i-\bar{x})^2}{n} \quad ; \quad s_y^2 = \frac{\Sigma(y_i-\bar{y})^2}{n}$$

$$r = \frac{\Sigma(x_i-\bar{x})(y_i-\bar{y})}{n \cdot \sqrt{s_x^2 \cdot s_y^2}} \quad ; \quad SE = \sqrt{s_y^2 .(1-r^2) \cdot \frac{n}{n\text{-}2}} = \sqrt{\frac{\Sigma(y-\bar{y})^2}{n\text{-}2}}$$

3. Formulae to calculate statistical features given in § 2 above.

$$\Sigma\,(x_i-\bar{x})^2 = n \cdot s_x^2 \quad ; \quad (y_i-\bar{y})^2 = n \cdot s_y^2$$

$$\Sigma(x_i-\bar{x})\,(y_i-\bar{y}) = n \cdot \sqrt{s_x^2 \cdot s_x^2} \cdot r$$

$$\Sigma(y_i-\hat{y})^2 = (n\text{-}2)\ s^2 \qquad \text{(residual square)}$$

4. Calculation procedures to estimate the regression constants:

$$\hat{Y}_i = bx_i + a$$

where x and y are log transformed;

$$b = \frac{n \cdot \sqrt{s_x^2 \cdot s_y^2} \cdot r}{n \cdot s_x^2} \quad ; \quad a = \bar{y} - bx\ .$$

5. Calculation procedures to estimate the regression constants without making the distinction between x and y required in regression analysis (orthogonal equation):

$$\hat{y}_i = bx_i + a,$$

where x and y are log transformed:

$$b = \sqrt{s_y^2 / s_x^2} \quad ; \quad a = \bar{y} - b\bar{x}$$

GLOSSARY

Symbols and units of measurements are also given for most abbreviations in Table 3.5.

1. Main Abbreviations:

j , i	inflow
o	outflow
λ	lake
e	euphotic
P	total phosphorus
PO_4–P	phosphate phosphorus
N	total nitrogen
Inorg-N, IN	inorganic nitrogen (NO_2 + NO_3 + NH_3)
Chl	chlorophyll *a*
Sec	Secchi disc transparency
PP	planktonic primary production
y	year
w	winter
sp	spring
s	summer
f	fall

2. Statistical Abbreviations:

n	number of data points
r	correlation coefficient
B	coefficient of determination
SE	standard error of estimate

3. Morphometry

A_0	lake surface area (L^2)
V_0	lake volume (L^3)
$\bar{z}$	mean depth (L^1)
Q_y	annual outflow from lake (L^3T^{-1})
T(w)	water residence time (V_0/Q_y , $\bar{z}/q_a$), (T)
q_a	hydraulic load (Q_y/A_0) , (TL^{-1})
r(w), ρ	flushing rate 1/T(w) , (T^{-1})

Examples:

Nutrient Loading

L(P)	specific loading of total phosphorus per unit lake surface area per year (ML^{-2} T^{-1});
$[P]_j$	average annual inflow concentration of total phosphorus ($L(P)q_a$) (ML^{-3})

$\frac{\overline{[P]}_j}{1 + T(w)}$	average annual concentration of total phosphorus in the lake, predicted from loading (ML^{-3}).

Nutrient Concentration and Biomass

$[P]_\lambda$	inlake concentration of total phosphorus at a given time (ML^{-3});
$\overline{[P]}_\lambda$	average annual concentration of total phosphorus in the lake (ML^{-3})
$\overline{[P]}_e$	average annual concentration of total phosphorus in the euphotic zone (ML^{-3})
$\overline{[chl]}$	average annual concentration of chlorophyll *a* in the euphotic zone (ML^{-3}).
$\begin{bmatrix} max \\ chl \end{bmatrix}$	annual peak concentration of chlorophyll *a* in the euphotic zone (ML^{-3}).

Other Measurements

$\overline{[Sec]}_y$	average annual Secchi disc transparency (L)
Σ PP	annual planktonik primary production per unit lake surface area ($ML^{-2}\ T^{-1}$).

Appendix 6

RELATIONSHIP BETWEEN INLAKE TOTAL PHOSPHORUS CONCENTRATION AND OUTFLOW CONCENTRATION IN THE OECD LAKES

The nutrient loading concept lakes are considered as mixed reactors which assumes that the lake concentration of phosphorus and the outflow concentrations are equal, $[P]_\lambda = [P]_0$ (see Appendix 1). Since this assumption is built into the empirical formulae used in the OECD Programme, it is instructive to examine this relationship, because any departure from this assumption would contribute an error to formula predictions.

The determination of the average lake concentration of a nutrient is seldom without difficulty. It requires an adequate number of sampling stations with adequate limnological skills and understanding. Up until now, however, surprisingly little attention has been paid to this basic problem. The relative ease or difficulty of estimating the average concentration in a water body is strongly influenced by the morphometry of the basin (i.e. size and shape), hydrology and the relative position of the major inflow(s) to the outflow. For example, if a major inflow or a major point source of phosphorus is very close to the outlet, a large proportion of the inflow water may leave the lake without adequate mixing in the basin. Under such conditions the estimated lake and outflow concentrations would be $\overline{[P]}_\lambda < \overline{[P]}_0$. Then, depending on the number and the location of the sampling sites upon which the inlake concentration estimate is based, the lake concentration predicted from loading $\overline{[P]}_j / \left(1 + \sqrt{T(w)}\right)$ would differ, more or less, from the average inlake concentration ($[P]_\lambda$).

Another extreme situation where the average lake concentration is difficult to estimate is in long, narrow and deep reservoirs where the inflow enters at the shallow end and the water is discharged at the opposite deep end. The Shallow Lakes and Reservoirs Report uses a theoretical example to illustrate the problem of estimating $\overline{[P]}_\lambda$ in such a reservoir, where inflow concentration of $\overline{[P]}_j = 100$ mg/m³ is assumed. If the water body is assumed to be a mixed reactor, by using Vollenweider's (1976) formula, the lake concentration would be 40 mg $\overline{[P]}_\lambda/m^3$, in that example. However, if the reservoir is considered as a series of four connected reactors, the volume weighted average of lake concentration would be 35 mg $\overline{[P]}_\lambda$, with values of 80 mg $\overline{[P]}_\lambda/m^3$ for the segment nearest to the inlet and 22 mg $\overline{[P]}_\lambda/m^3$ for the segment near the outlet. This would imply a $\overline{[P]}_0/m^3$ of 22 mg. Thus, in this situation $\overline{[P]}_\lambda > \overline{[P]}_0$, if the reservoir is treated as one single unit. Estimation of the average lake phosphorus concentration for a long, narrow reservoir is often based on the conditions near the outlet. In this case the average lake phosphorus concentration for the whole basin is underestimated, although it is made equal or nearly equal to the outflow concentration. The Kerr Reservoir in the USA, OECD Programme, is such an extreme example. The Nutbush Arm of the Kerr Reservoir has a total phosphorus inflow of 435 mg $\overline{[P]}_j/m^3$. The total phosphorus concentration is about 280 mg $\overline{[P]}_e/m^3$ at the shallow upper end; and it drops to 16 mg $\overline{[P]}_e/m^3$, at the deep end near the outflow. The latter figure then would be representative of the outflow phosphorus concentration. The average lake concentration estimated from loading using

Vollenweider's (1976) formula could be 134 mg $[P]_\lambda$, treating the basin as a unit, which is 4.5 times the lake concentration given in the USA Report. Consequently, the Kerr Reservoir Nutbush Arm appears as an outlier on a nutrient load-lake concentration plot.

These examples suggest that a water body of complicated morphometry or hydrology, should be treated as a series of segments for the purposes of nutrient load-trophic response relationships. By doing this, the average lake concentration in each segment would realistically represent the existing nutrient concentration and outflow concentration of that segment. Thus, a chain of water bodies would approximate conditions of a chain of mixed reactors. That treatment would reflect existing conditions more closely where the empirical formulae could be used successfully to predict nutrient load-trophic response relationships. Recently, Chapra (1978) used a similar approach when he treated embayments of large water bodies individually for the purposes of nutrient load-trophic response relationships.

As for the average lake concentration, estimation of outflow concentration is often subject to error. Disregarding problems associated with analytical determination of the instantaneous outflow concentration, an adequate frequency of sampling and an accurate hydraulic discharge estimate is required for accurate estimation of the outflow concentration.

The relationship between $[P]_0$ and $[P]_\lambda$, $[P]_e$ in the OECD Programme is examined separately in three groups: the combined OECD data, the Shallow Lakes and Reservoir Project and all natural water bodies grouped together (Alpine and Nordic Projects), (see Table A6-1, Figures A6.1 to A6.4). Estimates of $[P]_0$ concentrations are not available for the USA Project.

For the relationships examined, the correlation coefficients are high (r = .93 to .98), and they are highest in the Reservoir Project. Consequently, the standard error of estimates for the Reservoir Project (see .107 and .133) are considerably smaller than those found in the natural lakes (SE .182 and .183). The closer agreement between outflow concentration and inlake concentration in reservoirs may be explained by the fact that inlake concentrations are usually determined near the outflow, particularly in long and narrow reservoirs. Thus, the measurements of the average outflow concentration would reflect more closely the estimated average inlake concentration. This procedure, however, often seriously underestimates the true, average inlake concentration in the water body.

The slope of the regressions for the true relationships examined in the Reservoir Project is close to one and the intercepts are 1.06 and 1.19 for the $[P]_0 - [P]_e$ and $[P]_0 - [P]_\lambda$ relationships, respectively, (see Table A6.1, Figures A6.1 and A6.2). This means that both $[P]_\lambda$ and $[P]_e$ give very similar estimates of $[P]_0$, but in both cases the estimate of $[P]_0$ is about 10 per cent greater than the respective $[P]_\lambda$ or $[P]_e$ concentration. The higher $[P]_0$ concentration found in most reservoirs undoubtedly originates from the nature of these reservoirs, where nutrient-rich water is discharged from the hypolimnion during periods of thermal stratification.

In natural lakes, the slope of the regression of the $[P]_0 - [P]_\lambda$ relationship is 0.85 (see Table A6.1, Figure A6.3). This indicates that the increase in outflow phosphorus concentration is progressively slower with increasing inlake concentration. This decline in the rate of increase in outflow concentration with respect to inlake concentration, reflects the conditions found in eutrophic, stratified lakes where the phosphorus concentration is considerably higher than in the epilimnion. Consequently, the phosphorus concentration in the epilimnetic outflow is lower than the inlake concentration during periods of thermal stratification. For the same reason, the annual mean euphotic lake concentration approximates more closely the annual mean outflow concentration (Figure A6.4).

The relationships found in lakes were $[P]_0 \approx [P]_e$ and $[P] < [P]_\lambda$ confirm what limnological experience suggests. The good agreement found in these very simple relationships shows that the data base available in the OECD Programme, despiste its shortcomings, can be used to describe and predict average lake behaviour. However, the relatively simple relationship between $\overline{[P]}_0$ and $\overline{[P]}_\lambda$, and between $\overline{[P]}_0$ and $\overline{[P]}_e$ is not without uncertainties, which indicates that uncertainties will exist in the estimation of $\overline{[P]}_\lambda$, $\overline{[P]}_e$ and $\overline{[P]}_0$. *These errors undoubtedly would contribute to errors of prediction in the nutrient load-trophic response relationships.*

LIST OF FIGURES

Table A6.1

REGRESSION EQUATIONS,RELATING ANNUAL MEAN OUTFLOW CONCENTRATION OF PHOSPHORUS WITH ANNUAL MEAN INLAKE AND EUPHOTIC CONCENTRATIONS FOR THE COMBINED OECD DATA, SHALLOW LAKES AND RESERVOIRS PROJECT AND NATURAL LAKES

Project	Equation	r	SE	n	95% confidence limits	
					Intercept	Slope
Combined OECD data	$[P]_0 = 1.40\,[P]_\lambda^{.91}$	.95	.172	61	1.03 – 1.91	0.83 – 0.99
Combined OECD data	$[P]_0 = 1.20\,[P]_e^{.98}$	.96	.154	58	0.89 – 1.60	0.90 – 1.05
Shallow Lakes and Reservoirs	$[P]_0 = 1.19\,[P]_\lambda^{.99}$	.97	.133	25	0.81 – 1.76	0.89 – 1.08
Shallow Lakes and Reservoirs	$[P]_0 = 1.06\,[P]_e^{1.01}$	.98	.107	24	0.77 – 1.46	0.93 – 1.09
Natural Lakes (Alpine & Nordic	$[P]_0 = 1.64\,[P]_\lambda^{.85}$	.93	.183	36	1.08 – 2.51	0.73 – 0.96
Natural Lakes (Alpine & Nordic)	$[P]_0 = 1.32\,[P]_e^{.94}$	.93	.182	34	0.84 – 2.08	0.82 – 1.07

Figure A6.1

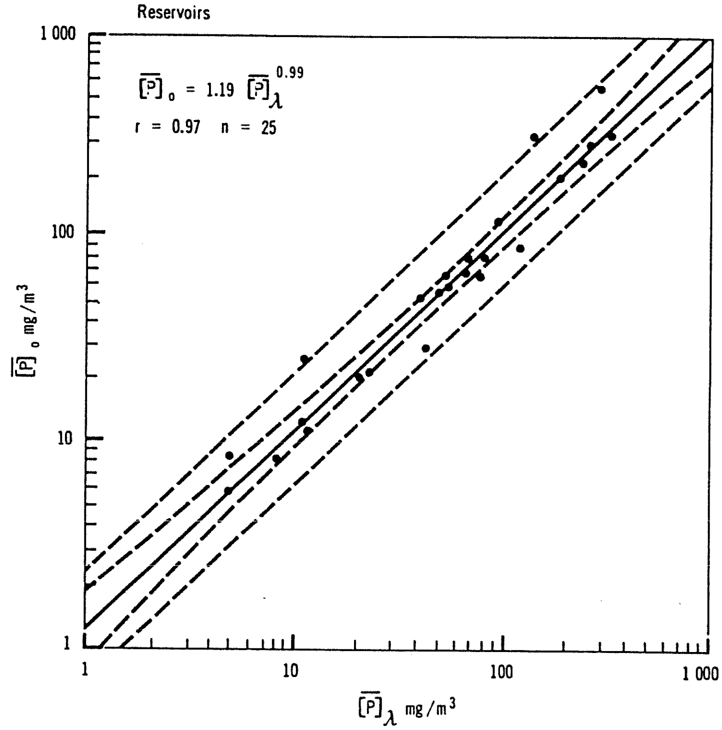


Figure A6.2

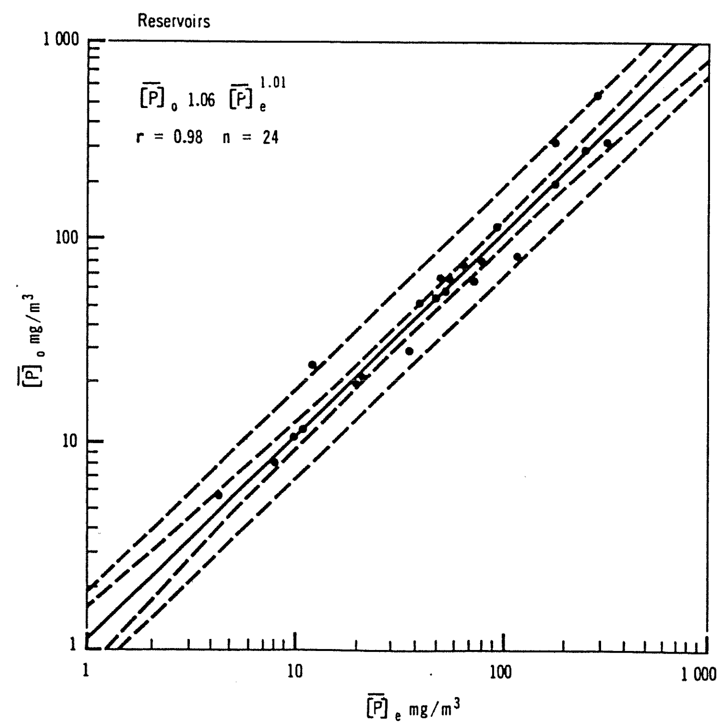


Figure A6.3

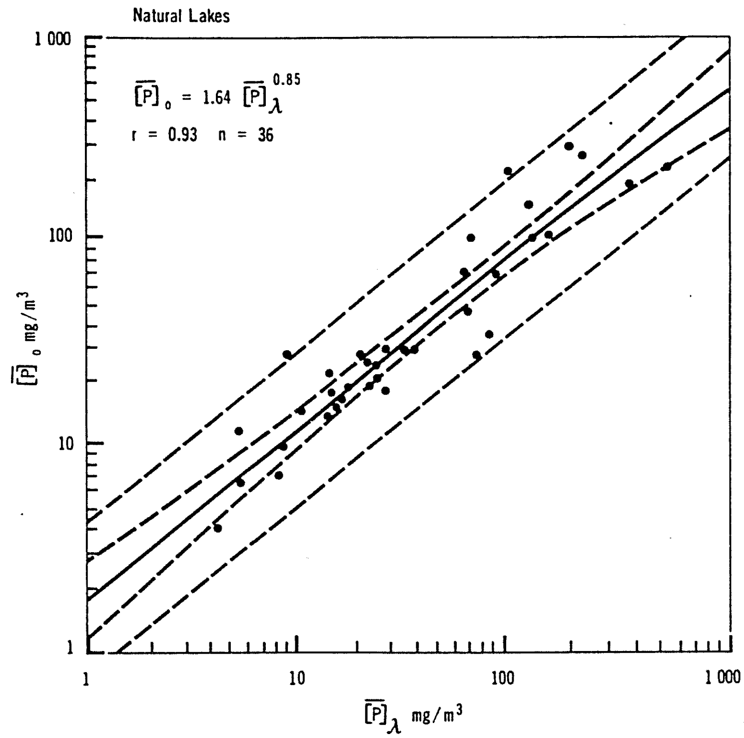
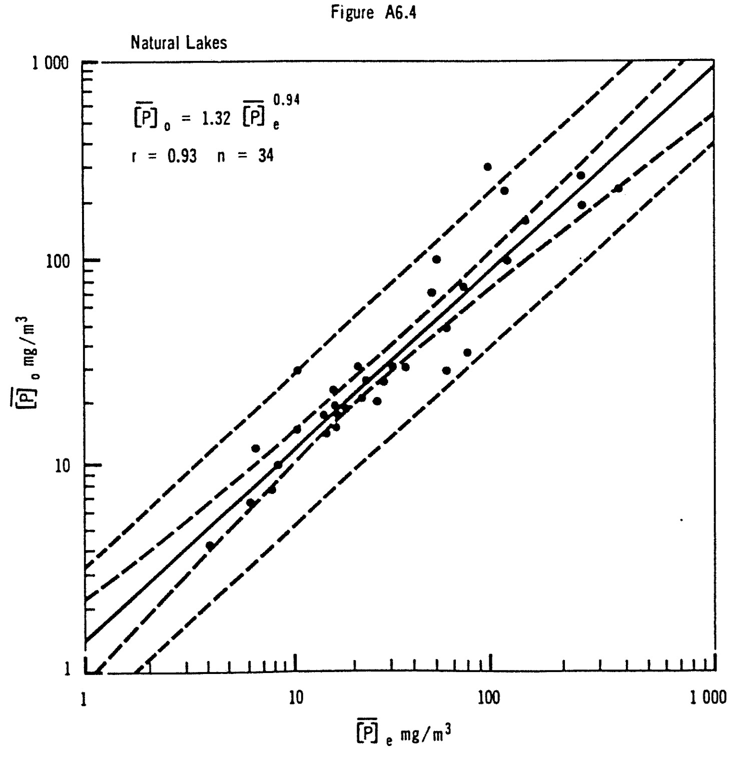


Figure A6.4



Appendix 7

CRITICAL ASSESSMENT OF SOME ASPECTS OF THE OVERALL INFORMATION TREATED IN THE OECD EUTROPHICATION REPORT

1. Data Base used in the OECD Eutrophication Programme

The success of the programme depended on well-coordinated monitoring projects. Therefore, great effort was devoted to the kind of variables measured, the selection of reliable practical analytical methods, sampling procedures and minimum sampling frequency. This was to ensure that adequate and comparable data could be obtained for later elaboration and analysis and that participants with relatively modest technical facilities could contribute.

Throughout the monitoring programme in 1975, the Technical Bureau issued guidelines to ensure uniform and comparable procedures for reporting the essential variables. These were revised in 1976 (OECD 1975, 1976).

It was stressed that several sampling stations were required to describe conditions in lakes with complex morphometry, but if this was not possible, the minimum provision was that the lake should be sampled at the deepest point (or points). Only this minimum provision was followed in many cases and often a distorted picture of the average lake concentration resulted. Guidelines were also given on the choice of depths at which to sample. It was proposed that during the period of stratification, samples are essential from above and below the thermocline and from lower down in the hypolimnion. Samples from the hypolimnion very close to the lake bottom were particularly important. An absolute minimum sampling frequency of four times per year was recommended (winter, summer, spring and autumn overturn) and a sampling frequency of at least once a month during periods of stratification. The frequency of sampling affects the various measurements differently. Infrequent sampling usually gives a distorted picture of the resultant variables which have short-term variability (Table 3.3) and it is inadequate for the determination of peak values of chlorophyll *a* and daily primary production.

The Technical Bureau also defined the units for the essential variables and clarified several uncertainties which arose during the workshps. The eutrophic zone was defined as the depth at which the light intensity of the photosynthetically active spectrum (400-700 nm) equals 1 per cent of the subsurface light intensity (from photometric measurements). Where this information is not available, a Secchi disc reading (in metres) in which z_e = 2.5 Secchi was used. The latter is of course only a rough estimation of the euphotic zone which may vary considerably, depending on the spectral composition (colour of the water). Calculation methods for annual mean and seasonal mean values were defined. The seasons were given as "winter, spring, summer and autumn". For water bodies showing irregular circulation patterns, it was recommended that breakdowns be made for two seasons only, "summer" and "winter". This made it possible to present the data in terms of annual means, which are essential for use in the nutrient loading formulae, while seasonal variation and seasonal peak values could still be recognised. Both these features are essential for understanding of the process of eutrophication.

Following these guidelines, the participants sent their data to the respective coordinating centres on a standard questionnaire and the centres prepared the data base for each project from this information. This was then returned to the participants for an accuracy check and for any corrections. This corrected data formed the basis for calculation of the computed variables, such as the water residence time, and inflow and outflow concentrations.

2. Weakness in the Data Base

The uniform treatment of data outlined above applies only to the European Projects, but even in the European Projects that were some inconsistencies. For example, in the Shallow Lakes and Reservoirs Project, annual median Secchi transparency was used while in the other projects mean values were used. The North American utilised on-going or already completed studies in which the guidelines of the Technical Bureau had not been followed. Consequently, the data base of the North American Project is less consistent and sometimes not comparable to the European Project. For example, total nitrogen and outflow concentrations of nutrients were not considered in the USA Project. Also, annual mean values for the variables considered were available only in a few cases, and where there were available, summer values were not usually given. Therefore, in the data analysis a mixture of mean annual, summer and ice-free season values had to be used in the North American Project.

The guidelines also covered the sampling and analytical methods to be used (Golterman and Clymo, 1969; and Vollenweider, 1969). The regional reports give details of the analytical methods and the sampling frequencies used in individual water bodies. During the monitoring programme, however, only the Nordic and Alpine Programmes conducted intercalibration studies. The Nordic intercalibration study was conducted at the very beginning of the programme, and so, participants were able to make any necessary improvements in methodology. Yet even within the Nordic Project(s) a number of laboratories did not employ the same methods as the rest. The intercalibration study of the Alpine Programme was conducted during the monitoring programme and the results became available towards the end of the programme. Thus, there was little opportunity to correct methodological errors. Intercalibration studies were not carried out in the Shallow Lakes and Reservoirs Project. In the absence of comprehensive quality control, the accuracy of any single data point is questionable, and there is no doubt that some of the variability observed in the data analysis is caused by measurement errors.

The meaning of the value of a variable is another source of innacuracy. The sampling frequency varied greatly among the projects, thus, a mean value may represent four or less measurements, while in some cases it was obtained from a year-round weekly sampling. Depending on the conditions that exist in a water body, infrequent sampling might give adequate information for calculation of a mean value; this is true for most pristine oligotrophic lakes. It could, however, give a rather distorted picture in a highly eutrophic lake with a seasonally variable nutrient load.

An obvious fundamental weakness of the data base is that only in a few cases were all the recommended variables measured. Nitrogen an primary production is missing in a large number of cases, while in a few cases, even total phosphorus concentration is not available. This means that the number of data points varies considerably in the various relationships examined and some of these relationships are not directly comparable.

3. Key Variables Monitored

3.1. Nutrients

It became apparent at the beginning of the monitoring phase of the programme that for all projects including the North American Project which only included on-going studies, only a very few participants would be able to measure all the recommended fractions of phosphorus (i.e. total-P, particulate-P, total soluble-P, soluble "ortho" phosphate PO_4-P) and nitrogen (particulate organic-, ammonia-nitrate-, nitrate- and soluble organic nitrogen). Therefore, the Technical Bureau decided that only total phosphorus and ortho phosphate (PO_4-P) fractions and total nitrogen and the mineral dissolved nitrogen fraction (NO_3-N, NO_2-N, NH_3-N) should be measured for nutrient loading and nutrient concentrations. Even this limited objective could not be met by several participants and data, particularly for nitrogen loading and lake concentrations, are missing in a few cases. With the exception of some participants in the Nordic Project, who used Koroleff's (1969) persulphate oxidation method, most investigators used either the Kjeldahl method or the photo-oxidation method (Golterman et al. 1978) for the determination of total nitrogen. The Kjeldahl method includes organic N and NH_3^-—N (not NO_2^-—N) while the persulphate oxidation and the photo-oxidation method are equivalent to Kjeldahl nitrogen plus NO_2^-—N and NO_3^-—N. The Kjeldahl method is rather imprecise at very low concentrations, and therefore, several investigators of oligotrophic waters not having access to photo-oxidation equipment or being in aware of the persulfate-oxidation method, were unable to provide adequate information on total N.

It has already been mentionned that total phosphorus and not other phosphorus species, was selected as the key variable in the Programme for practical rather than theoretical reasons. The various forms of phosphorus are defined in terms of the method used for their determination rather than by their biological significance. Therefore, because of the relatively simple methodology for total phosphorus determination and the increasing available data on it, total phosphorus was recommended for monitoring purposes.

From the outset it was recognised that the problems of the biological significance of total phosphorus, and of phosphorus availability will have to be resolved to understand fully the nutrient load-trophic response relationships. Phosphorus availability has been studied by several authors (e.g., Vollenweider, 1968; Lean, 1973; Rigler, 1974; Schaffner and Oglesby, 1978). Schaffner and Oglesby, 1978 considered that total phosphorus includes some or all of the following fractions: crystalline, occluded, absorbed, particulate organic, soluble organic and soluble inorganic phosphorus. Out of these fractions they defined "biologically available phosphorus" which includes soluble reactive phosphorus, soluble unreactive phosphorus and "labile" phosphorus; the three biologically available phosphorus fractions are listed in order of decreasing availability. Soluble reactive phosphorus is a mixture of dissolved inorganic and organic species and is measured by the method of Murphy and Riley (1962). This fraction is considered to be entirely biologically available. Some of the soluble unreactive phosphorus includes dissolved phosphorus fed by persulfate oxidation, also considered to be available for phytoplankton by enzimatic hydraulisation which frees organically bound fractions. Labile phosphorus is associated with soil particles and dissolves into an aqueous solution as determined by the sorption isotherm method of Tayler and Kunishi (1971). Undoubtedly, the use of biologically available phosphorus is an improvement on total phosphorus as a key variable in phosphorus load - trophic response relationships. However, the term biologically available phosphorus still remains somewhat vague because it describes a mixture of phosphorus fractions of different availability. Vollenweider (1979), by means of a simple diagram, indicated the order of phosphorus availability at the molecular level in relation to trophic response, and also the sources which should be considered as priorities in nutrient control measures.

DIAGRAM SHOWING RELATIVE IMPORTANCE OF PHOSPHORUS AVAILABILITY AT THE MOLECULAR LEVEL AND ACCORDING TO SOURCES

A) At the molecular level

PO_4 Inorganic polyphosphates — Low molecular organic P — High molecular organic P — Apatite-P; Fe, Al compounds

Mostly through enzymatic hydralisation

B) According to sources

Urban sewage — Erosinal runoff

+ certain industrial effluents — Leaching from forests and agricultural areas

Increased biological availability of phosphorus

The pathways of phosphorus and nitrogen through a water body are fundamentally different. Phosphorus may enter a water body through the inflows, precipitation, dry fallout and from the sediments, and it may be removed by sedimentation and through the outflow. Nitrogen has a more complex pathway. In addition to the inputs and outputs described for phosphorus, nitrogen can enter and leave a water body in the form of free nitrogen gas (N_2^-) through atmospheric exchange. Dissolved molecular N_2^- of atmospheric origin can be used by nitrogen fixing blue-green algae when the N:P ratio falls and nitrogen is in short supply and excess nitrogen can be removed from the inorganic nitrogen compartment by bacterial denitrification through the reduction of NO_3 to N_2 which may return to the atmosphere.

The different pathways of phosphorus and nitrogen in lake metabolism make phosphorus the obvious choice for eutrophication control. On the one hand phosphorus cannot enter a lake from the atmosphere as easily as nitrogen and on the other hand a certain reduction of phosphorus input will generally result in a greater reduction in algal biomass compared with the same reduction of nitrogen. Furthermore, the reduction of nitrogen input without a proportional reduction in phosphorus, creates low N/P ratio which favours nitrogen fixing nuisance algae, without any reduction in algal biomass.

3.2. Chlorophyll a

Chlorophyll *a* was chosen as the principal variable to use as a trophic state indicator. From a theoretical point of view, primary production which can be related to nutrient supply is the best trophic state indicator, but its adequate measurement is not without difficulties. It requires frequent samplings and a rather elaborate methodology, which is available only to a limited number of investigators. However, there is generally a good agreement between planktonic primary production and algal biomass, and algal biomass is an excellent trophic state indicator. Furthermore, algal biomass is associated with the visible symptoms of eutrophication, and it is usually the cause of the practical problems resulting from eutrophication. The usefulness of chlorophyll *a* as an indicator of algal biomass is widely accepted by the vast majority of aquatic scientists both for fresh and salt water. Chlorophyll *a* is relatively easy to measure compared to algal biomass and this measurement could be provided by almost all participants. The methods used for chlorophyll *a* determination in the programme are corrected for phaeophytin, the pigment fraction which is not active in photosynthesis. This improved methodology gives a more meaningful basis from which to establish chlorophyll *a*: total phosphorus ratios than can be obtained from the chlorophyll data used (i.e. Sakamoto - 1966 -). It could be argued, however, that the uncorrected (for phaeophytin) chlorophyll data used by Sakamoto might be better for chlorophyll total phosphorus correlations, since paheophytin also contains phosphorus.

One serious weakness of the use of chlorophyll *a* to represent algal biomass is the great variability of cellular chlorophyll content depending on algal species. Nicholls and Dillon (1978) showed from a literature survey that the chlorophyll content of algae can range over from 0.1 to 9.7 per cent of fresh algal weight. Radiation intensity and nutrient availability, particularly nitrogen, appear to be major factors affecting the chlorophyll content of algal cells. Although chlorophyll *a* reflects algal biomass quite well as a whole, a great variability in individual cases can be expected, either seasonally or on an annual basis due to a species composition, light conditions and nutrient availability.

3.3. Hypolimnetic Oxygen Depletion and Annual Primary Production

In addition to chlorophyll *a*, planktonic primary production and hypolimnetic oxygen depletion rates were obtained as trophic state indicators where they were available. In contrast to daily rates of primary production which have a very high short-term variability and are difficult to measure, hypolimnetic oxygen depletion has a low short-term variability and is relatively easy to measure. However, hypolimnetic oxygen depletion measurements can be obtained in deep lakes only, which eliminates a large number of shallow lakes from consideration. Similarly, annual primary production also has a low short-term variability, and thus, is extremely useful as a trophic state indicator, but it is very difficult to measure.

Annual primary production may be estimated adequately from a large number of daily primary production rates, while hypolimnetic oxygen deficit can be obtained from two sets of measurements taken during the period of summer stratification. Consequently, values for annual primary production are available for very few lakes in the project but lakes with data on hypolimnetic oxygen deficit are somewhat better represented.

Anaerobic hypolimnetic conditions caused by overfertilisation are one of the undesirable effects of eutrophication. Therefore, from a scientific and practical point of view, it is desirable to relate nutrient load to hypolimnetic oxygen conditions. The hypolimnetic oxygen conditions reflect the intensity of primary production of a lake by indicating the rate of use of the oxygen required for the composition of organic matter produced in the trophogenic zone and reaching the hypolimnion. Hypolimnetic oxygen depletion rates can be considered as a desirable index of lake productivity; Hutchinson and Mortimer (see Hutchinson, 1957) considered the use of areal oxygen deficit to separate lakes with respect to oligo- and eutrophy. To avoid erroneous conclusions concerning trophic state, lakes with high inputs of allochthonous organic matter or lakes where water colour is over 10 pt. units, should not be used for oxygen deficit calculations.

The hypolimnetic areal oxygen deficit was calculated as the difference between the hypolimnetic oxygen content at the onset and at the end of summer stratification, or in highly eutrophic lakes until just before oxygen depletion takes place in the hypolimnion. Apparently, in the Alpine Project for several highly eutrophic lakes where the oxygen depletion in the epilimnion develops quickly after the onset of summer stratification, the theoretical oxygen demand was calculated by totalling the amount of reduced substances, taking into account Mn, Fe, H_2S, NH_3, NO_2, CH_4

and particulate N. Unfortunately, the lakes for which these calculations were performed were not identified. For oxygen deficit calculations, the hypolimnion was considered only for lakes with a well-defined thermocline (> 1 C/m) at the end of the summer stratification. It was defined as beginning downwards from the depth of the inflection point during the two months preceding the onset of the fall overturn. Only lakes where the hypolimnetic to epilimnetic volume ratio is at least 1.5 were considered.

Care was taken with lakes in which metalimnetic oxygen maxima and minima occurred and in lakes where the euphotic zone extended into the hypolimnion. Quite late in the programme in 1976, it became apparent that unit volume hypolimnetic oxygen deficit would also be desirable for the final analysis. A request was made to this effect to the various projects without complete success. Only the Alpine Project provided this information.

3.4. Secchi Disc Transparency

At the end of the monitoring phase of the programme during the Shallow Lakes and Reservoirs Project workshop in 1976, it became apparent that the size of the Secchi disc was not uniform throughout the Programme. Also, some investigators used an entirely white disc while the majority used the standard 20 cm disc with black and white quadrants (Welch, 1948). It is also not clear whether a water telescope was used or not. Furthermore, some investigators failed to indicate the presence of water colour and inorganic turbidity. An effort was made for this report to remove from analysis the Secchi disc data where colour or inorganic turbidity was present, but it is possible that data which should have been removed, could not be identified. The combination of these problems certainly reduces the value of the analysed results which depend on the Secchi disc transparency data.

The problems associated with the Secchi disc measurements suggest that a sizeable number of investigators are strongly influenced by the traditions and habits of the region in which they work. Some are reluctant to change or modify established methods because often they do not wish to break the continuity with previous work which was often very high quality. Or very often they are unaware that the methods they use are somewhat different to those of others. They may not read new manuals and recommendations carefully and therefore do not realise that their own methods are not identical with what is recommended and used by the majority of investigators. Furthermore, students may learn these differing methods at one place of work without ever checking the origin and detailed description of these methods. Thus, the diversity is perpetuated which in turn prevents the total acceptance of a uniform system of methodology and measurements even in areas where it is possible and desirable. The problem which apparently exists with the Secchi disc transparency measurements, which is perhaps one of the oldest and simplest of all measurements employed in limnology, illustrates the urgent need to *maintain* a uniform system of measurements and methodology wherever it is possible and desirable.

3.5. Nutrient Loading

An understanding of the role of nutrient loading in lake productivity was the main objective of the programme and special emphasis was placed on the nutrient load estimation procedures.

To obtain an accurate nutrient load estimate, all sources must be considered (Vollenweider and Dillon, 1974), as well as the sediment to water flux (internal load). The role of precipitation is particularly important for lakes with small drainage area compared to lake surface and aeration. Internal loading cannot be measured directly and therefore, for the OECD Eutrophication Programme, only external loading estimates were requested. It was recognised that loading estimates are only accurate to ± 25%, even under ideal conditions and that reduced frequency of sampling increases the error and uncertainty in nutrient load calculations (Treunert et al. 1974).

Detailed information about the methodology used in estimating the nutrient budgets for the OECD Eutrophication Programme is given in the regional reports. Since the nutrient budgets are the very centre of the programme, it is essential to know the quality and validity of the nutrient load estimates. The evaluation of loading estimates is a difficult task because there are no absolute standards and one has to rely on accepted empirical data such as watershed phosphorus export coefficients to evaluate loading estimates. Another possible approach is to compare nutrient load estimates with predictions from nutrient load models, but this is not without pitfalls.

Rast and Lee in the USA Report attempted to evaluate the validity of the nutrient load estimates reported by the U.S. OECD investigators by using various approaches. First, they plotted the mean total lake phosphorus/mean inflow phosphorus concentration ratio ($\overline{[P]}_\lambda/\overline{[P]}_j$) against a hydraulic residence time expression, $1/(1+\sqrt{T(w)'})$ derived from Vollenweider's (1976) model. A major deviation of $\overline{[P]}_\lambda/\overline{[P]}_j$ from $1/(1+\sqrt{T(w)'})$ would make a reported loading suspect. In a second attempt they plotted the investigator's loading estimate against phosphorus loading calculated from published phosphorus export coefficients. Finally, they plotted phosphorus load estimated from Vollenweider's model $L(P) = \overline{[P]}_\lambda.(q_a\ (1+\sqrt{T(w)'})$ against phosphorus loading calculated from export coefficients to check whether use of the export coefficient data is justified for the purpose described above. They found a good agreement between the two variables on the latter plot. Most lakes fell between the lines representing ± 1:2 ratios on the plots which means that for most lakes the phosphorus load was not over or under-estimated by more than twofold. They considered this as a good agreement and that both the Vollenweider model and the phosphorus export coefficient data may be used to evaluate the reasonableness of the loading estimate. Rast and Lee defined a reasonable phosphorus loading as one which is within a factor of two (± twofold) on the first two plots discussed above. They justified this definition by stating that the standard deviation of the relative error of $1/(1+\sqrt{T(w)'})$ as the reference value, corresponds closely with the ± 2 x assumption. Subsequently, in their report they gave special consideration and scrutiny to lakes that fell outside the ± 2 x lines on those plots but they stressed that if a lake was outside the ± 2 x lines, it did not necessarily mean that the loading estimate was incorrect. It could also mean that the phosphorus cycling in that lake is different from that of most other water bodies (discussion of loading evaluation of other projects will follow).

Investigations in the Alpine Project revealed that the use of export coefficients requires great caution to avoid erroneous loading estimates. Phosphorus export coefficients obtained in a certain area cannot be applied to other areas without critical evaluation.

3.6. General Limitation of the Data Base

Throughout the programme, correlation techniques were used to test variables that may be relevant in the nutrient load – lake concentration – trophic response relationships. Also, empirical formulae derived from mass balance equations were used for the prediction of trophic responses to nutrient loading and vice versa. These formulae were developed using empirical data and their limits of prediction are restricted to the range of the data sets used in the formulae (Reckhow, 1978; Chapra and Reckhow, 1979). Therefore, the geometric means, the extremes and number of observations for the most commonly used variables in the Programme are presented in Table 3.5. to indicate the limitations of formulae and relationships examined. It is also clear from Table 3.5 that the number of observations for each variable differs greatly. Consequently, relationships based on a different number of data points and on a different range of values, are not directly comparable and when such relationships are compared, great care should be exercised to avoid erroneous conclusions.

3.7 Remarks on the Monitoring Programme

One can conclude that in spite of some shortcomings and uncertainties, the data gathered during the monitoring programme, is on the whole, of sufficient quality to form the basis of a valid and credible correlation analysis, and that it has achieved what can be expected realistically from such a large and diverse cooperative programme. The data base is sufficiently large and well-defined, and it encompasses a wide range of geographic, trophic and morphometric conditions. The information available is broad enough to establish the general statistical behaviour of lakes with respect to nutrient load and trophic response which represents most types of lakes in the major regions of Europe and North America. It should be noted, however, that subtropical (in USA) and Arctic lakes (including high Alpine) are poorly represented, and saline, closed basin lakes are not represented at all in the programme. Because the data is well documented, it is considered adequate in most cases to determine the general behaviour of lakes and to identify aberrant behaviour of individual water bodies.

Overall, the monitoring phase of the programme can be considered a success. The momentum initiated by the International Biological Programme in 1964 was maintained and a common system of measurements was developed and successfully put into use. The system of measurements which was used in this programme, although its emphasis was on variables that closely relate to eutrophication, can be viewed as a major contribution toward the goal of a common system of measurements used in limnology. The monitoring programme also stimulated the exchange of ideas between the participants and opened direct communication between various institutes in many countries of three continents. This is one of the major successes of the programme.

4. The Limiting Nutrient

It is of primary interest to examine the nitrogen and phosphorus relationship in the OECD lakes. The role of these nutrients in algal productivity is well documented (Sawyer, 1947, Vollenweider, 1968; Goldman et al., 1972; Chiadani and Vighi, 1974; Vallentyne, 1974; Schindler, 1977) and it was reviewed by Rast and Lee (1978). From the point of view of eutrophication, control of the relative proportions of phosphorus and nitrogen is particularly important. It is generally accepted that on average, the tissues of phytoplankton and aquatic macrophytes contain phosphorus and nitrogen atoms in the ratio of 1P:16N (by atom) and the plants require these nutrients in this proportion for growth. If the N:P ratio in the water is $>$ 16, this would mean that phosphorus atoms are unsufficient for algal growth and thus algal biomass is limited by the quantity of phosphorus present. Nitrogen limitation is present when the N:P atomic ratio falls below 16. The 1P:16N atomic ratio represents 1P:7.2N weight ratio per 500 wet weight of algae. This means that under ideal conditions the addition of phosphorus can produce 500 times its weight in the form of plant material, while the addition of nitrogen can only produce 69 times its weight.

However, care should be taken in applying the N:P ratios from the literature. Often the conditions under which results were obtained are not describes adequately, and very often the biological availability of phosphorus is not known which could lead to a low N:P ratio. Consequently, the various N:P limiting ratios given in the literature differ somewhat. Forsberg et al. (1978) summarised the various ranges of N:P ratios giving the approximate ratios which indicate whether N or P or both are the limiting nutrient (Table A7.1). This summary table can be used as a guide to determine the type of nutrient limitation in lakes.

Table A7.1

APPROXIMATE LEVELS OF NITROGEN AND PHOSPHORUS LIMITATION AS INDICATED BY N:P RATIOS IN LAKES

Total nitrogen / Total phosphorus		Dissolved nitrogen / Phosphate phosphorus		Limiting nutrient
by weight	in moles	by weight	in moles	
< 10	22	< 5	10	N
10-17		5-12		N and/or P
> 17	37.6	> 12	26.5	P

There is no sharp boundary between phosphorus and nitrogen limitation. Sakamoto (1966) suggested in a survey of a large number of Japanese lakes that for lakes with N:P ratios between 9 and 17 (by weight), the nitrogen and phosphorus concentrations are nearly balanced with the ratio required for the growth of phytoplankton. He considered lakes with > 15 – 17 N:P ratios phosphorus limited and below 9–10 N:P ratios, nitrogen limited. Using mainly Sakamoto's data, Dillon and Rigler (1974) considered N:P ratios of 12 where the lakes are phosphorus limited. Chiadani and Vighi (1974) found in a study of the relative nutrient requirements of algae that phosphorus is limiting at about 10N:1P (by weight) and nitrogen at below 5N:1P, and the limiting relationship was proportional between these values. Schindler (1877) found that phosphorus-proportional development of phytoplankton was maintained in fertilised lakes with N:P ratios as low as 5:1-through the fixation of atmospheric N_2. He noted that the low N:P ratios favour nitrogen fixing, buoyant blue-green algae which are most undesirable from the water quality view point.

4.1 The $\overline{[N]}_\lambda - \overline{[P]}_\lambda$ Relationship

The relationship between $\overline{[N]}_\lambda$ and $\overline{[P]}_\lambda$ in the OECD lakes is presented in Figure 4.1 and the N:P ratios of 7:1 and 15:1 (by weight) are indicated with dotted lines. The ~ 7N:1P ratio represents the condition around and below which nitrogen limitation can be assumed, and above the ~ 15N:1P ratio phosphorus limitation is assumed. Between the two lines the water body may be limited by either N or P or by both. A relatively good correlation ($r = .75$) exists between the two variables. The regression line shown crosses the line indicating the 7N:1P ratio at very high phosphorus concentrations (~ 800 mg P), suggesting an increasing tendency to nitrogen limitation in extreme eutrophic conditions. This obviously suggests that high lake nutrient concentrations are caused by excessive loading of nutrients with low N/P ratios in highly eutrophic lakes. Sewage effluents contain a high ratio of inorganic phosphorus to inorganic nitrogen (1P:4N by weight) and other elements required for aquatic plant growth and this results in change from phosphorus limitation to nitrogen limitation in the receiving water body (Vallentyne, 1974). The resulting shift in N/P ratios favours nitrogen fixing, buoyant blue-green algae which are most undesirable from a water quality view point. Schindler (1977) found similar good correlation between $\overline{[P]}_\lambda$ and $\overline{[N]}_\lambda$ at a lower range of concentrations (up to 40 mg $\overline{[P]}_\lambda/m^3$) in lakes where he conducted fertilisation experiments. He pointed out that the nitrogen concentrations also increased in lakes which received additional phosphorus input with little or no nitrogen addition. He concluded that the additional nitrogen was obtained by nitrogen fixation.

In the OECD lakes where nitrogen data are available, five lakes fell near the line for nitrogen limitation which suggests at least seasonal nitrogen limitation (Greifensee, Hallwillersee, Léman, Lough Neagh, Pusiano). However, this assumes equal availability of nitrogen and phosphorus. If the total phosphorus determined for some or all of these lakes is only partially biologically available, these lakes then could be phosphorus limited. Only one lake, Lake Esrøm, can be considered as nitrogen limited according to the N:P ratios in the OECD lakes.

4.2. The Inorganic Nitrogen – Inorganic Phosphorus Relationship

For the purposes of algal growth, nutrients are required in the inorganic form. Thus, the use of inorganic nitrogen and phosphorus ratios is often considered more meaningful in determining which nutrient is limiting, than the total N and P ratios. The relationship between $[\overline{\text{Inorg N}}]_\lambda$ (NH_4^+, NO_2^-, $NO_3^- - N$) and $[\overline{PO_4 - P}]_\lambda$ concentration for the OECD lakes is shown in Figure 4.4. with an indication of the 7N:1P and 15N:1P ratios (by weight). Out of the 89 water bodies where data is available, 18 fell below or near the 7N:1P ratio which suggests that these lakes are nitrogen limited, at least seasonally. Nitrogen and phosphorus can alternate seasonally as the limiting nutrient and this can change spatially along the length of long morphometrically complex lakes where mixing is incomplete and the inflow concentrations differ greatly from the average lake concentrations.

5. Unsolved Problem Areas

5.1. Shortcoming of the Phosphorus Loading Models and the OECD Data

The empirical phosphorus loading models considered are derived from the mass balance model which assumes steady state conditions, and that there is no change in concentration in the water column and no internal loading. The models also assume that the lake concentration and the outflow concentration are equal. In fact, a lake is seldom at a steady state on a daily basis even if the nutrient and hydrologic load are reasonably constant from year to year. If the nutrient supply is changing due to an increase in population, industrial activity or use of fertilisers and detergents, the lake will obviously not be at a steady state (Vollenweider, 1968, Dillon and Rigler, 1974; Chapra and Tarapchak, 1976).

The evidence available in the literature appears to support the assumption that on an annual basis the average lake concentration equals the outflow concentration (Reckhow, 1978). However, the accurate estimation of average lake concentrations is often difficult, particularly in large lakes and in lakes of complicated morphometry (El-Shaarawi and Shah, 1978). It is also particularly complicated when a local inflow concentration is appreciably greater than the average lake concentration ($[\overline{P}]_j \gg [\overline{P}]_\lambda$). Under such conditions, local imbalances occur ("local reactors") which interfere with the precise calculation and interpretation of the average lake concentration (see Appendix 6).

The empirical phosphorus models used in the OECD Eutrophication Programme are based on a mixed reactor model and ignore the internal lake mechanisms. It was necessary to over-simplify some of the essential relationships such as the interactive, dynamic several-layer systems. Statistically obtained parameter connections were employed when necessary. According to Vollenweider (1976), this approach can be defended partly because of our present inability to cope adequate by with complete interactions between different system components in different lakes, particularly because some of the interactions are non-linear and stochastic in nature. Therefore, on occasions, more or less defendable shortcuts were employed to reach solutions for rather complex interactions. Thus, the nutrient supply and trophic response were oversimplified to a great extent, but it was done deliberately in order to clarify these problems.

One of the fundamental features of the empirical phosphorus loading models is the sedimentation component function (i.e., s(P), $v_s(P)$ and Rp) which is essential to all models and cannot be measured directly. To overcome this difficulty, the models employ a statistical approximation of the sedimentation coefficient of phosphorus $\left(s(P)\right)$ by using apparent settling velocity values of phosphorus $\left(v_s(P)\right)$ as a constant. The settling velocity is called "apparent" to indicate that it is an oversimplification – an artifact of the model's structure and time scale – because a phosphorus atom is subject to many transformations and a range of transport mechanisms before it is incorporated into the sediments (Chapra, 1977). The apparent settling velocity describes net in-lake losses of phosphorus on an annual basis. The sedimentation coefficient of phosphorus can be determined empirically for a given lake, subject to the uncertainty associated with the measurements of phosphorus load, but it is known to vary from lake to lake. Even from a cursory examination of the plot used by Vollenweider (1976) when he determined the average sedimentation coefficient of phosphorus used in his models, it is apparent that individual values varies greatly (up to several times) compared to the mean value of $10/\bar{z}$, which is equivalent to an average apparent settling velocity of 10 m/y. Similarly, the retention coefficient of phosphorus (R_p) varied considerably at a given water discharge height (q_a) on the graphical representation of the model which was used by Chapra (1975) and Dillon and Kirchner (1975) to estimate average apparent settling velocities ($v_s(P) = 16.0$ and 13.2 m/y respectively) used in phosphorus loading models. While the average apparent settling velocities used in the models operate well when used for a large spectrum of lakes, individual lakes with higher or lower than average rates of phosphorus sedimentation (i.e. diatoms, phosphorus absorption to inorganic particles, or extremely buoyant phytoplankton) are expected to differ from model predictions. The difficulties associated with the adequate estimation of the sedimentation coefficient is only one of several uncertainties associated with the phosphorus loading models.

It must be stressed that in spite of a concentrated effort, there is uncertainty about the accuracy of many data points obtained from a relatively limited number of observations in the OECD Eutrophication Programme. This is particularly applicable to the North American Project which used already available data often gathered for other purposes. Applicable to all data is the uncertainty reflecting the natural variability of a given situation, particularly loading. Treunert et al. (1974) and Kleiber and Elebach (1978) have shown that decreasing sampling frequency increases the uncertainty in loading estimates. On this basis alone, it is unlikely that the accuracy of individual loading estimates is better than ± 25 per cent. In addition to this, annual variations in loading have been found to be at least of the same order of magnitude and in some cases appreciably higher, even in undisturbed natural systems (Schindler et al. 1976). Thus, the reported loading estimates have a built-in uncertainty of at least ± 35 per cent. Similar considerations apply to other variables such as nutrient concentrations, chlorophyll, planktonic primary production, etc. The uncertainty associated with lake concentrations is further complicated by the difficulties of estimating average lake concentrations for basins exhibiting a peculiar shape or incomplete mixing, factors previously mentioned. Similar uncertainties apply, in most cases, particularly in shallow lakes and reservoirs, to the hydraulic (i.e. T(w) and morphometric (i.e. $\bar{z}$) variables used in the models. Thus, a non-negligible error in model prediction can be expected to stem from the error of estimation of the variables employed in the model data set.

5.2. Some Reservations Concerning the Uncritical Use of the Nutrient Loading Models

The central aim of the OECD International Cooperative Programme for the Monitoring of Inland Waters was to examine the trophic response to nutrient loading of a spectrum of lakes representing a wide range of geographic, hydraulic and trophic conditions and develop an optimal lake management strategy for the purpose of controlling eutrophication. The nutrient loading concept employed in the programme makes it possible by using simple correlation techniques to generalise on the average statistical behaviour on lakes in response to nutrient loads. The programme confirmed that a quantifiable relationship exists between the drainage basin as exemplified by the intensity of phosphorus supply and the receiving body of water. If this relationship is expressed in quantitative terms, it can be used to attempt to manage lake systems by enabling estimation of the nutrient supply reduction required for lakes which are considered overfertilised for their intended use. Statistically, it is possible to predict the average lake phosphorus concentration and the average standing stock of phytoplankton in terms of chlorophyll *a* concentration and annual primary production. By using the same relationships, it is possible to identify outliers, that is, water bodies which respond differently, and the reasons for their deviating behaviour (which could not have been detected otherwise) can be studied.

The participants were aware throughout the programme that the trophic response (real or apparent) to nutrient load is modified by a complex interaction of morphometric, physical, chemical and biological factors. They were also conscious of the numerous assumptions made in the simple nutrient loading models used in the programme and also that some of these assumptions were often not met. Therefore, it is stressed that the generalisations made in this programme should not be used rigidly and blindly. It should be kept in mind that biological factors alone could profoundly modify both the nutrient load and the real or apparent trophic response in a lake. These factors should also be considered before a costly treatment facility, which would not give the expected results particularly in the shorter term, is implemented.

In the following section, the important assumptions used in the nutrient load - trophic response models are listed, along with factors known to modify the trophic response in water bodies, in order to facilitate the recognition of possible causes of unexpected trophic response.

The phosphorus loading models are derived from the mass balance model which assumes:

- steady state conditions in a completely mixed reactor;
- no change in concentration in the water column;
- that the lake concentration and the outflow concentration are equal;
- that internal loading is not present;
- that the sedimentation component fraction, which is an essential component of all phosphorus loading models, cannot be measured directly. Thus, statistical approximations are employed in the various models which may deviate considerably (up to several times) in some individual cases;
- that the nutrient load estimate is accurate (it is unlikely that individual loading estimates are more accurate than ± 35%);
- that nutrient load expressed in terms of annual rate reasonably approximates steady state conditions;
- that nutrient load is at a steady state on an annual basis, (this is seldom the case, even in undisturbed natural systems, considerable year to year variations in nutrient load were detected due to fluctuations in annual runoff);
- that the basin is open and that there is an annual water surplus or outflow from the lake (the models cannot be applied without modification to closed basins or arid and semi-arid regions);
- that these models are to be used in phosphorus limited lakes only.

5.3 Some Sensitive Areas where Limnological Knowledge is Essential in Applying the Results of the OECD Eutrophication Programme

- The models employ total phosphorus which includes all phosphorus fractions present but some of this phosphorus is not available for plant growth; the percentage of biologically available phosphorus may vary considerably in individual cases and regions.
- Phosphorus which is bound to soil particles (apatite) and is not biologically available, should not be used in phosphorus loading calculations.
- In some coloured, humic lakes, a large proportion of the total phosphorus is not biologically available.
- Macrophytes and filamentous algae are ignored in the models; macrophytes may contain large amounts of phosphorus which is ignored in the determination of $\overline{[P]}_\lambda$.
- Macrophytes often act as nutrient pumps and may cause appreciable internal loading.
- In anoxic, eutrophic lakes, large and usually unknown quantities of phosphorus are released from the sediments; this is not taken into account in the loading calculations and may result in a greater $\overline{[P]}_\lambda$ and a higher trophic response than predicted from loading (see A7.5.3).
- Biological activity such as by bottom feeding fish and emerging bottom dwelling invertebrate fauna often produces considerable internal loading of nutrients.
- The presence or absence of fish and types present in a lake can profoundly affect the apparent trophic response; in the absence of predation highly abundant zooplankton reduce phytoplankton by grazing which results in lower than expected chlorophyll *a* concentrations:
- The type of algal community can profoundly affect the sedimentation rate of phosphorus; lakes dominated by diatoms remove phosphorus by sedimentation at a much faster rate than lakes dominated by more buoyant green or blue-green algae.
- The chlorophyll *a*—total phosphorus ratio, an index of the efficiency of algae to utilise phosphorus varies greatly among lakes (about 0.05 to 0.7 in the OECD programme). A shift in algal composition and the efficiency of phosphorus use can profoundly affect the trophic response in a given water body.
- In very large water bodies and those of complex morphometry (not "mixed reactors") the realistic determination of average lake concentration representative for the basin may require a large number of measurements.
- Concentration measurements from a single station, usually at z_{max} are often not representative for the basin.
- In large water bodies or those complex morphometry, localised imbalances in nutrient concentration and trophic response may develop ("local reactors"). In extreme cases localised symptoms of eutrophication may develop near point sources of nutrients, in an otherwise oligotrophic water body.
- The alteration of nutrient load is expected to alter the algal composition in a water body which in some instances can profoundly alter the rate of sedimentation of phosphorus and the efficiency of use of phosphorus by algae; thus the intensity of trophic response may also deviate from expected behaviour depending on the algal species present, particularly in the short term, until the water body reaches a new equilibrium.
- In shallow unstratified lakes, the recycling and utilisation of nutrients is more efficient than in deep stratified lakes.
- In reservoirs, peculiar flow regimes and hypolimnetic water withdrawal (rich in nutrients) should be taken into account.

- Because of the reasons listed above and their interactions, the reduction of phosphorus load may not have the expected result of a similar reduction in trophic response, as would be expected from the average statistical behaviour of water bodies.
- The nutrient loading models give an estimation of average conditions; local conditions may deviate considerably, temporally and/or spatially.
- The prediction of lake concentration should be given with statistical confidence limits to reflect more realistically the uncertainties in the model prediction.
- The empirical phosphorus loading model should be used only for water bodies within the range of conditions (hydrologic, morphometric, trophic and climatic) represented by the model development data set.

5.4. Internal Loading

Where anoxic conditions develop at the water-sediment interface, substances contained in the sediments, including nutrients, are realised into the water column (Mortimer, 1941, 1942). Vollenweider (1968) and Lee (1970) pointed to the importance of mixing processes within sediments and between sediments and the overlying waters in controlling the release of phosphorus and other substances from the sediments. It is now widely recognised that the mixing processes which occur between sediments and the overlying water play an important role in the overall phosphorus release from the sediments under both oxic and anoxic conditions (Lee et al., 1977; Ryding and Forsberg, 1977; Schindler et al., 1977). This seems to confirm that internal loading of phosphorus is a more serious threat in shallow lakes than in deep lakes (Vollenweider, 1968). The importance of mixing processes in the recycling of nutrients into lake waters was stressed further by Fee (1979) when he showed that rates of planktonic primary production during the ice-free season expressed per unit volume of the epilimnion, are linearly related to the ratio of epilimnion sediment area in unfertilised lakes.

In extreme cases, internal loading of phosphorus could exceed that of external loading (Vollenweider, 1968). When heavy internal loading exists, the observed lake concentration of phosphorus can far exceed the value estimated from external loading only. This condition is prevalent when the external load is reduced after a long history of accelerated eutrophication. Under such conditions the lake concentration $[P]_\lambda$ not only exceeds the concentration estimated from the external load $[P]_{\lambda L}$ but it can exceed that of the average inflow concentration $[P]_j$ (e.g., Baldegersee, Greifensee, Kreutzrichter Basin of Lucerne).

Experience gained in various recent lake restoration schemes suggests that the history of accelerated eutrophication, that is, the length of time the lake has been eutrophied, has an important bearing on lake behaviour with respect to internal loading and phosphorus retention in the sediments (Edmonson, 1972; Michalski and Conroy, 1973; Larsen et al., 1975; Schindler, Schindler et al., 1973, 1977; Ryding and Forsberg, 1977; Ahlgren, 1977). Apparently the history of eutrophication and the amount and type of sediments accumulated, determine the time required for recovery of the lake after the external load is reduced. Lake morphometry further modifies the lake response to external loading. In shallow lakes where water circulation is rapid and substances are quickly released from the epilimnion sediment, the internal load would be more severe than in deep lakes where most of the substances released from the anoxic hypolimnion sediment are prevented from entering the epilimnion during periods of thermal stratification.

Schindler et al. (1977), showed that in artificially eutrophied lakes, with a previous oligotrophic history, 81 per cent of total phosphorus and 61 per cent of total nitrogen added over a six-year period were deposited in the sediments. In addition 1 and 11 per cent, respectively, appeared in the water column and the remainder was lost through the outflow. Despite the increased total phosphorus content in the sediment and the anoxic conditions, the interstitial water contained less than 1 mg/m^3 molybdate-reactive phosphorus, which is typical of the natural oligotrophic lakes in the area. By contrast, Ryding and Forsberg (1977) observed that after reduction of the external phosphorus load, in three shallow highly eutrophied lakes with a long history with eutrophication 22 to 400 per cent of the external phosphorus load was released from the sediments. The net release of phosphorus under oxic conditions in the lake exceed that of the rates obtained in the laboratory under anoxic conditions. They suggested that mixing induced by the wind force plays a dominant role in the transport of nutrients from the sediments of shallow lakes. In the three lakes after the phosphorus load reduction, the lake concentration $[P]_\lambda$ was much higher than that estimated from the external load $[P]_{\lambda L}$ and the amount of phosphorus loss through the outflow was greater than this external load. They also found heavy internal loading of nitrogen, but a large proportion of the nitrogen (36 - 58 per cent) was lost to the atmosphere as the result of denitrification. Ahlgren (1977) showed that during six years after the reduction of the external phosphorus load in a deeper, thermally stratified lake, the decrease of lake phosphorus concentration followed a simple dilution curve. During the same period, the yearly loss of phosphorus through the outflow was greater (more than two times higher in two years) or nearly equal to the external phosphorus loading. In the later years net deposits of phosphorus occurred on an annual basis, but seasonally, summer releases were sufficient to maintain a heavy algal bloom.

Experience gained from lake recovery studies suggests that the condition of the sediments is important in the extent of eutrophication and in the recovery of lakes. Sediments remain oligotrophic and only become gradually eutrophic, long after the water mass becomes highly eutrophic (Schindler et al., 1977). Conversely, the highly eutrophic sediment would remain eutrophic long after the external load is reduced and would thus delay the recovery of the lake.

A better understanding of the quantitative aspects of the release of phosphorus from the sediments is needed in order to predict with confidence the time required for the recovery of a heavily eutrophied lake.

5.5. Toxic and Potentially Hazardous Substances

The OECD Eutrophication Programme was established to clarify the role of nutrients in eutrophication; it dit not examine the role of toxic and potentially harmful substances associated with accelerated eutrophication. It is recognised, however, that along with an increased trophic response, other harmful effects of certain substances are part of the overall problem of man-made eutrophication.

Population growth and increased economic activity not only increase the nutrient load to water bodies but also increase the loading of other substances which may be hazardous to aquatic life and to human health. Some of these substances such as trace elements were always present in low quantities in aquatic systems supplied in the basic natural load, but with accelerated eutrophication, the increased amounts supplied, accumulated and recycled in the aquatic system cause problems. Other substances, mainly organic compounds of an anthropogenic nature, originating from pesticides, paints and other chemicals, also enter into water courses and add to the problem. These substances are usually found in very low concentrations in water but they can accumulate in animal tissues and persists in a water body.

a) Trace Elements:

Mercury, lead, arsenic, cadmium, selenium, cooper, zinc, chromium, and vadium could cause serious local problems near point sources of industrial releases. The additive and synergistic effects of the mixture of heavy metals can further increase the hazard to aquatic life. Mercury and lead rank highest with respect to real or anticipated environmental hazard (Anon., 1978). Both of these elements can be converted by the process of methylation by microorganisms into methyl mercury and methyl lead, which are strong human nerve poisons.

b) Organic Compounds:

Organochlorine pesticides such as DDT, Aldrin-dieldrin, chlordene, polychlorinated biphenyls (PCBs), are of particular environmental concern. These organic compounds are extremely persistant chemicals and have the ability to bioaccumulate. These substances are known to cause reproductive failure in fish-eating birds, either by failure of eggs to hatch or by the production of non-viable ofsspring.

c) Microorganisms:

Pathogenic organisms can enter water systems from direct sewage discharge, sewer overflows and septic tank failures. Depending on the size of the water body, they can cause health hazards in nearshore regions or they can affect the whole water body.

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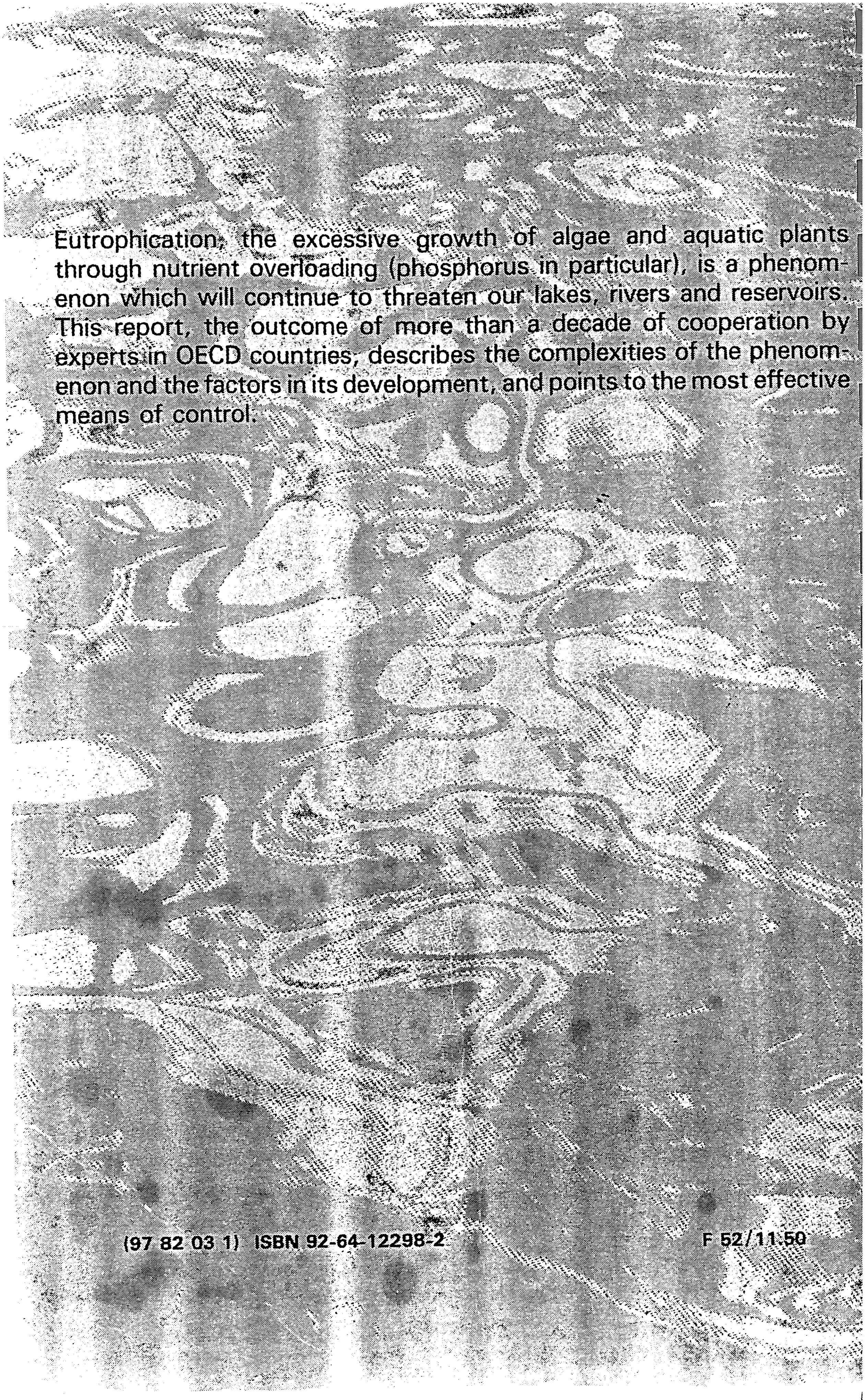

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