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SUMMARY ANALYSIS OF THE NORTH AMERICAN (U.S. PORTION) OECD EUTROPHICATION PROJECT: Nutrient Loading—Lake Response Relationships and Trophic State Indices

Environmental Research Laboratory
Office of Research and Development
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SUMMARY ANALYSIS OF THE NORTH AMERICAN (US PORTION) OECD EUTROPHICATION PROJECT: NUTRIENT LOADING - LAKE RESPONSE RELATIONSHIPS AND TROPHIC STATE INDICES

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report provides an extensive examination of relationships between nutrient inputs and lake responses and, therefore, should be extremely valuable to those people concerned with lake management and controlling accelerated lake eutrophication.

A.F. Bartsch
Director, CERL

PREFACE

Several years ago the Organization for Economic Cooperation and Development (OECD) member countries, including the USA, initiated a eutrophication study with the primary objective of formulating the relationships between aquatic plant nutrient loadings to lakes and impoundments and the response of these water bodies to these loadings. Emphasis was on the development of relationships that could be used to identify critical aquatic plant nutrient (i.e. nitrogen and phosphorus) loadings in order to avoid or minimize water quality problems caused by excessive fertilization (eutrophication). In the majority of the participating countries, the OECD eutrophication study caused the initiation of field studies, using the same or similar sampling techniques and analytical methods, to assess aquatic plant nutrient loadings to a water body and its response to these loadings. In the US, however, the lack of funds to initiate comparable studies of US water bodies limited the United States' participation in the overall study. The US EPA did, however, provide small grants to enable investigators who had already conducted nutrient load-response studies in US water bodies to develop a report of their studies which emphasized nutrient load-lake response relationships in accord with overall OECD Eutrophication Program objectives and format. Funds were also provided by the US EPA to prepare this summary report. This report represents an initial analysis of the results of the US portion of the North American Project of the OECD eutrophication study.

The goal of the OECD eutrophication study is the quantification of the relationships between nutrient loading and trophic response in lakes and impoundments. Attention in this initial analysis has been focused mainly on evaluation of the nutrient loading portion of this relationship, especially as these nutrient loadings are related to the critical nutrient loading levels and the trophic response of the US OECD water bodies, using the Vollenweider phosphorus and nitrogen loading diagrams. This report also evaluates the nutrient sources, nutrient budget calculation methodologies, and nutrient loading estimates reported by the US OECD investigators for their respective water bodies. The US OECD water body nutrient loadings have been evaluated several ways, including: (1) several relationships developed by Vollenweider, (2) comparison with calculated nutrient loadings based on watershed nutrient export coefficients and land usage patterns within the watershed, and (3) other nutrient loading-lake response relationships developed by Vollenweider, Dillon, and Larsen and Mercier. In addition, an attempt was made in this summary report to formulate some of the relationships between nutrient load-lake and impoundment water quality responses, based on the data available for the US OECD water bodies.

This report also presents a discussion of the application of the US OECD eutrophication study results for predicting the changes in water quality that will arise from altering the phosphorus input to lakes and impoundments. The US OECD water bodies are ranked in accord with various previously proposed trophic status index systems. A new trophic status index system based on a modification of the Vollenweider phosphorus loading relationships is presented. A modified Vollenweider phosphorus loading relationship has been developed which enables individuals concerned with water quality management to select the appropriate phosphorus loadings for achieving a desired level of chlorophyll, water clarity, and hypolimnetic oxygen depletion rate.

Upon completion of this study a copy of those sections of the report pertinent to each investigator's water bodies was sent to the investigators and a request was made for them to review and comment on these sections. Approximately half of the US OECD eutrophication study investigators responded to this request. In the two years from the time that the US OECD eutrophication investigators had provided the data which served as the basis of this report and the completion of this report, several investigators have done additional work on their respective water bodies. The new data was brought to the authors' attention as part of the review process. In most cases the changes in the data were relatively minor and did not change the conclusions of the report. In others, major changes in the nutrient loads for their water body were reported, under conditions where the investigator indicated that the new data more reliably estimated the nutrient loads and should be used instead of the ones reported previously.

All suggested changes of the investigators have been noted in this report and in the appendices. Major changes have been used as a basis for rewriting sections of this report. This situation will cause differences between the data presented in the investigator's report published as a companion volume, and the data presented in this report.

ABSTRACT

The US participation in the OECD Eutrophication Program consisted of having 20 investigators prepare reports on the nutrient load-lake and impoundment response relationships for their respective water bodies. This report presents a critical review of these overall relationships with particular emphasis given to evaluation of the Vollenweider nutrient load-trophic state formulations. This review includes consideration of the nutrient load response relationships for 38 water bodies, or parts of water bodies, located throughout the US, with the preponderance located in the northern half of the US. It has been found that the Vollenweider nutrient load relationship involving water body mean depth, hydraulic residence time and phosphorus load correlates well with the trophic states assigned by the US OECD eutrophication study investigators.

A good correlation has also been found between phosphorus loading, normalized as to hydraulic residence time and mean depth, and the average chlorophyll and water clarity (as measured by Secchi depth) for the US OECD water bodies. In general, phosphorus and nitrogen loads to US OECD water bodies were within a factor of ± two of the loads predicted on the basis of average nutrient concentrations within the water bodies and on the land use patterns within the water body watersheds. Generalized nutrient export coefficients have been developed in this study, enabling estimates of nutrient loads to be made on the basis of land use patterns within the watershed.

The relationships developed in this study can be used to predict the improvement in water quality that will result from a change in the phosphorus load to a water body for which phosphorus is the key chemical element limiting planktonic algal growth. The US OECD water bodies all show approximately the same trophic status when evaluated by several recently-proposed trophic state index systems. A new trophic state index system has been developed in this study which is based on the relationship between the actual phosphorus loading and permissible phosphorus loading as defined in the Vollenweider phosphorus loading and mean depth/hydraulic residence time relationship. This relationship has been modified to enable water quality managers to determine the appropriate phosphorus load for a particular water body in order to yield a certain chlorophyll content from planktonic algae and its corresponding water clarity. It is recommended that these relationships be used as a basis for

establishing critical phosphorus loads to lakes and impoundments.

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This report is essentially the same as the Ph.D. dissertation of Walter Rast for The University of Texas at Dallas.

SECTION I

INTRODUCTION

The excessive fertilization (eutrophication) of natural waters is one of the most significant causes of water quality deterioration in the US and in many other countries. This increasing eutrophication, resulting principally from the cultural activities of man, is occurring because of the excessive input of aquatic plant nutrients into water bodies. Some water bodies are naturally eutrophic in that they receive sufficient supplies of aquatic plant nutrients, mainly phosphorus and nitrogen, from natural sources to produce excessive growths of algae and macrophytes. However, many of man's activities which accelerate this transport of aquatic plant nutrients into water bodies can greatly accelerate the eutrophication process. While eutrophication may be desirable in some water bodies to increase productivity, in general the eutrophication process is associated with water quality deterioration. Excessive algal or macrophyte growths can result in a significant deterioration of water quality, which can greatly hinder the waters' use for domestic and industrial water supplies, for irrigation and for recreation. Today eutrophication ranks as one of the most significant causes of water quality problems in the US, and it will probably become of greater concern as other water pollution problems are alleviated (Lee, 1971).

While other elements have occasionally been proposed (Goldman, 1964; Provasoli, 1969; Kerr et al., 1970; Schelske and Stoermer, 1972), phosphorus and nitrogen are generally considered to be the major nutrients controlling or limiting the productivity of water bodies, and hence the eutrophication process. Of these two nutrients, the key element most often found limiting aquatic plant populations is phosphorus (Vollenweider, 1968; Lee, 1971; 1973; Vollenweider and Dillon, 1974). Furthermore, in many instances, phosphorus inputs to water bodies are from point sources such as domestic wastewaters. By contrast, large inputs of nitrogen are frequently from non-point (diffuse) sources such as agricultural runoff, precipitation, dry fallout and nitrogen fixation. These diffuse sources are usually more difficult to control. In general, phosphorus inputs are often more amenable to control measures than are nitrogen inputs (Vollenweider and Dillon, 1974). Water bodies which are normally nitrogen-limited can possibly be made phosphorus-limited if the phosphorus inputs are reduced sufficiently.

Eutrophication control is frequently based on limiting aquatic plant nutrient inputs, usually phosphorus. Attempting to control the eutrophication process by controlling phosphorus inputs to natural waters is both technically sound and economically feasible for many water bodies (Lee, 1973; OECD, 1974a; Vollenweider and Dillon, 1974). However, such a strategy requires that the relationships between the phosphorus inputs and the trophic responses of the aquatic plant populations of a given water body be understood on a quantitative basis. Development of such an understanding has always been an extremely difficult problem because the eutrophication process is a complex physical, chemical and biological phenomenon (Sawyer, 1966; Fruh et al., 1966; Fruh, 1967; Stewart and Rohlich, 1967; Vollenweider, 1968; Federal Water Quality Administration, 1969; National Academy of Science, 1969; Lee, 1971; 1973; Likens, 1972a; US EPA, 1973a).

It has not been possible in the past to quantitatively relate the phosphorus loading of a given water body to the resultant aquatic plant related trophic response, as reflected in its relative degree of eutrophication. Consequently, the management of water systems subjected to cultural eutrophication has been largely subjective. Extensive, and often expensive, programs of aquatic plant nutrient removal from domestic wastewaters or diversion of point source inputs of nutrients have been initiated in an attempt to alleviate eutrophication problems in lakes and impoundments. These programs have no quantitative data on the expected effects of these programs on trophic response and water quality in these water bodies. Clearly, a quantitative methodology is required to initiate effective water quality management with some assurance that the desired results will be attained.

In an attempt to alleviate this situation, the Organization for Economic Cooperation and Development (OECD) member countries initiated the Cooperative Programme for the Monitoring of Inland Waters, which was designed to provide quantitative data on the aquatic plant nutrient load-lake and impoundment response relationships, with particular emphasis on water quality and the development of approaches to be used for water quality management of excessive fertility problems.

SECTION II

CONCLUSIONS

1. Based on the initial analysis of the US OECD eutrophication study data, the approach developed and modified by Vollenweider, relating the phosphorus loading of a phosphorus-limited water body to its morphological and hydrological characteristics, has considerable validity as a method for determining critical phosphorus loading levels and associated overall degree of fertility for US lakes and impoundments.

2. The findings of this initial analysis give considerable support to the recent adoption of the Vollenweider nutrient loading-water body fertility response relationship by the US EPA as a basis for establishing phosphorus loading water quality criteria.

3. Initial analysis of the US OECD data indicates the Vollenweider phosphorus critical loading criteria, developed for water bodies located in northern temperate climates, also appears to be applicable to warm climate water bodies such as those found in the southern and southwestern US. Additional study needs to be done on water bodies in this region to confirm this preliminary conclusion.

4. The Vollenweider phosphorus critical loading criteria, developed for planktonic algal responses to phosphorus loadings, will likely have to be modified in order to be applicable to water bodies whose primary productivity and aquatic plant nuisance problems are manifested mainly in macrophyte and attached algal growth. Modifications of the critical phosphorus loadings will likely be required where the primary problem arising from the excessive fertility is domestic water supply water quality. Further, it is possible that the Vollenweider approach will not be applicable to impoundments with hydraulic residence times in the order of a month or less, and especially for those impoundments that show marked stratification of inflowing waters during critical growing seasons.

5. The results of this study indicate the feasibility of using the Vollenweider approach for determining critical nitrogen loading levels and trophic state associations for nitrogen-limited water bodies.

6. The similar relative positioning of the US OECD water bodies

on both the phosphorus loading and nitrogen loading versus mean depth/hydraulic residence time diagrams illustrates the relatively constant ratio of nitrogen to phosphorus loading to water bodies.

7. The relationship developed by Vollenweider, between a water body's phosphorus loadings and its mean influent phosphorus concentration and hydraulic loading, as well as the use of watershed land use nutrient export coefficients, appear to be effective means for determining the reasonableness of the phosphorus and nitrogen loading estimates to a water body.

8. The trophic relationships developed by Vollenweider, between a water body's phosphorus loading characteristics and its mean chlorophyll concentration; by Dillon, between phosphorus loading and phosphorus retention coefficient and mean depth; and by Larsen and Mercier, between mean influent phosphorus concentration and phosphorus retention coefficient, also appear to be potential tools for estimating phosphorus loads, average phosphorus content and associated overall degree of fertility for many US lakes and impoundments.

9. Because of the lack of uniform analytical and sampling methodologies, direct comparisons of eutrophication data between the US OECD water bodies must be made with caution. In general, the correlations between phosphorus loading-concentrations and eutrophication response data are better than those observed between nitrogen loading-concentration and the same response parameters, and support the observations of phosphorus-limitation of most of the US OECD water bodies.

10. The water quality models derived from the relationships between phosphorus loading and chlorophyll $\underline{a}$, phosphorus loading and Secchi depth and phosphorus loading and hypolimnetic oxygen depletion offer simple, practical and quantitative methodologies for assessing the expected effects of eutrophication control programs based on phosphorus removal from domestic wastewaters and other phosphorus control programs, on water quality in the affected water bodies.

11. The recently proposed trophic status index systems of the US EPA, Carlson, and Piwoni and Lee produce relatively similar trophic rankings for the US OECD water bodies, suggesting that their common ranking parameters may equate their trophic ranking abilities.

12. The trophic status index system based on excess phosphorus loading and excess chlorophyll $\underline{a}$, derived in this report, offers promise as a trophic ranking system based on the phosphorus loading and expected water quality responses in water bodies.

13. The Vollenweider phosphorus loading and mean depth/hydraulic

residence time diagram can be related to the common water quality parameters of chlorophyll _a_, Secchi depth and hypolimnetic oxygen depletion, based on the relationships between total phosphorus, chlorophyll _a_, Secchi depth and hypolimnetic oxygen depletion in natural waters.

SECTION III

RECOMMENDATIONS

1. The US EPA and the states should adopt the modified Vollenweider phosphorus load and mean depth/hydraulic residence time relationship for determining the permissible phosphorus loading for phosphorus-limited lakes and impoundments where the primary concern is the impairment of water quality for recreational use. The recently proposed US EPA Quality Criteria for Water (US EPA, 1975b) should be modified to include this recent modification of Vollenweider's model, as well as the approaches presented by Dillon, and Larsen and Mercier.

2. The US should continue to actively participate in the international OECD Eutrophication Program data review, synthesis and report preparation. Such participation is likely to result in a much better understanding of the types of water bodies that obey the modified Vollenweider nutrient loading relationship.

3. Research funds should be made available at the federal and state levels to further investigate the applicability of the Vollenweider nutrient loading relationships for lakes and impoundments located in the southern half of the US as well as for water bodies with high levels of inorganic turbidity, color, attached algae and macrophyte, and floating macrophyte water quality problems. Also, special consideration should be given to water bodies with short hydraulic residence times and shallow depths and to impoundments which show high degrees of stratified inter or underflow waters.

4. Studies should be conducted to further refine the permissible versus excessive loading criteria, giving particular attention to differences in water quality problems associated with recreational use in various regions of the US, especially the southern half of the US, and the critical nutrient loadings for impairment of domestic water supply water quality.

5. Further work should be done to establish a relationship between the critical phosphorus loading relationship as defined by Vollenweider, the actual phosphorus loading for a given water body, and its associated water quality. The ultimate

objective of these studies should be the development of quantitative relationships which can be used to further predict a change in the water body's water quality as a function of an altered nutrient load. Particular attention should be given to assessment of water quality in terms of planktonic algal growth, attached algae and macrophyte growth, chlorophyll concentration, water clarity and hypolimnetic oxygen depletion.

6. Studies should be conducted to develop similar nitrogen relationships and information as described above for phosphorus.

7. Studies need to be conducted to examine the significance of utilizing total phosphorus and total nitrogen as a basis for establishing loading criteria versus using the available forms of these nutrients for establishing loading criteria.

SECTION IV

ORGANIZATION FOR ECONOMIC COOPERATION AND DEVELOPMENT

The Organization for Economic Cooperation and Development (OECD) is an independent, international organization headquartered in Paris. It is concerned primarily with the economic growth of its twenty-four member nations. These comprise the more highly developed countries of the world, excluding the Communist-bloc nations. As a group, they produce more than 60 percent of the world's wealth and enjoy the world's highest per capita incomes (OECD, 1973a; 1974b). The member nations are presented in Table 1.

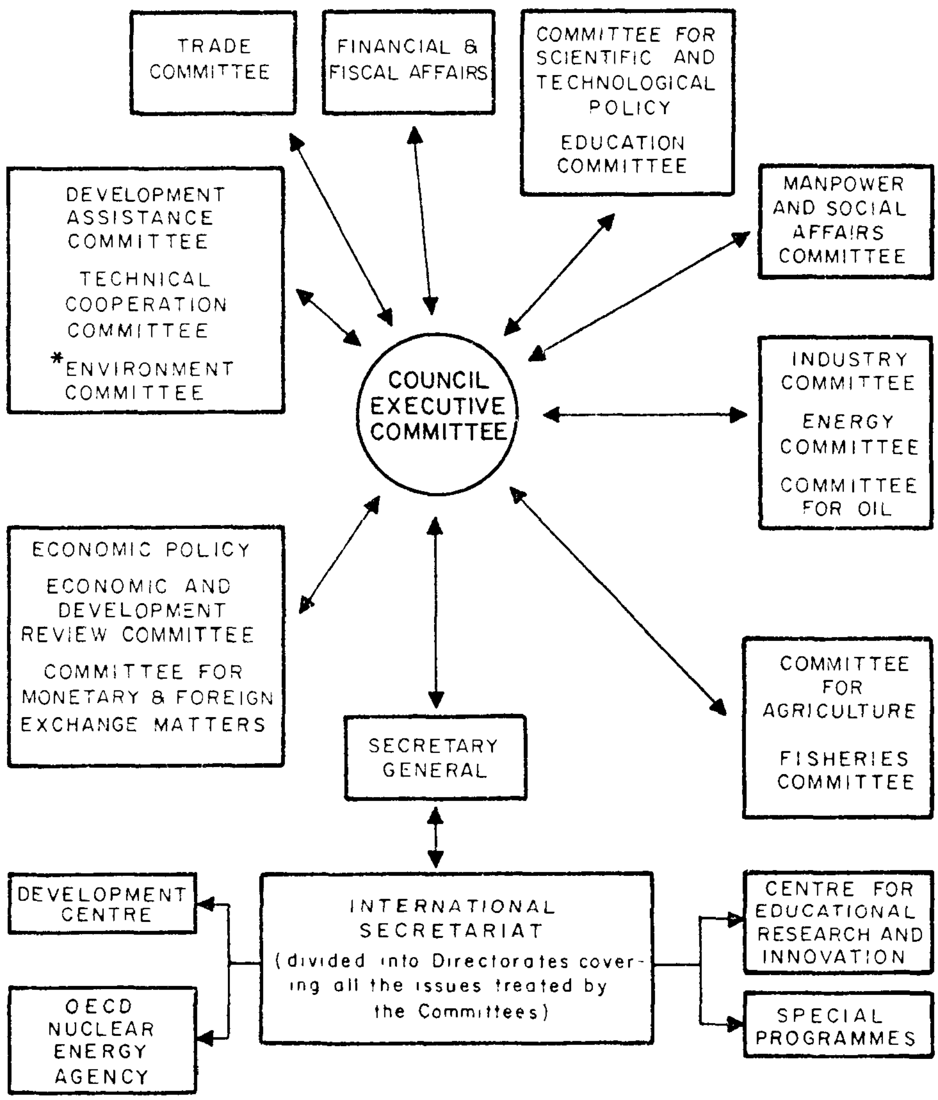
Table 1. OECD MEMBER COUNTRIES

Australia	Greece	Norway
Austria	Iceland	Portugal
Belgium	Ireland	Spain
Canada	Italy	Sweden
Denmark	Japan	Switzerland
Finland	Luxembourg	Turkey
France	Netherlands	United Kingdom
Germany	New Zealand	United States
Special Status Country: Yugoslavia		

(From OECD, 1973a)

Because economic development of the member nations is its organizational focus, OECD contains a number of committees associated with the various aspects of economic development and growth. These committees and the OECD organizational structure are presented in Figure 1. Recognizing that economic productivity frequently gives rise to environmental problems, the OECD has concerned itself with both the quantitative and qualitative aspects of economic development. In 1970 it transformed its Committee for Research Cooperation into the more comprehensive Environment Committee, which is responsible for:

1. investigating the problems of preserving or improving man's environment, with particular reference to economic and trade implications;



*Formerly the Committee for Research Cooperation
(From OECD, 1973a)

Figure 1. Organizational Structure of OECD.

2. reviewing and confronting actions taken or proposed in member nations in the field of environment, together with their economic trade implications;

3. proposing solutions for environmental problems that would, as far as possible, take into account all relevant factors including cost effectiveness; and

4. insuring that the results of environmental investigations can be effectively utilized in the wider framework of the Organization's work on economic policy and social development.

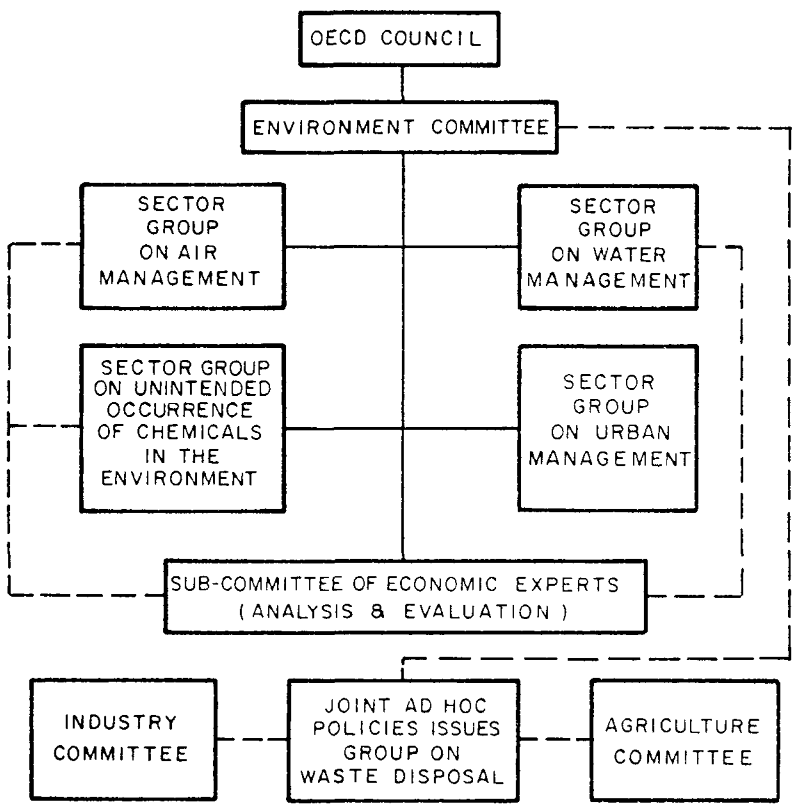
The Environment Committee is assisted by various delegate groups concerned with the development of policy in specific areas of overall environmental problems. These delegate groups are presently concerned with the environmental problems of water and air pollution, automobile and aircraft noise, traffic congestion and urban transport and hazardous chemicals (OECD, 1973a; 1974a). The Environmental Committee and its associated delegate groups are outlined in Figure 2.

WATER MANAGEMENT SECTOR GROUP

Concern over the problems of decreased water quality caused by eutrophication had been expressed by OECD even before the formation of the Environment Committee. Eutrophication of various degrees of severity had been observed in lakes, flowing waters and impoundments in most of the world's highly developed nations for many years (Vollenweider, 1968). An ad hoc group of the OECD Committee for Research Cooperation, chaired by O. Jaag (EAWAG, Zurich), recommended that a study be made of the existing literature on eutrophication, with particular reference to the roles of phosphorus and nitrogen in the eutrophication process. This study, completed by R.A. Vollenweider, resulted in the 1968 report, "Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters With Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication" (Vollenweider, 1968). This report noted the lack of "sufficient relevant measurement data" for producing precise guidelines for eutrophication control.

In 1967, the Water Management Research Group was formed. In May, 1968, this group held a symposium in Skokloster, Uppsala, Sweden on large lakes and impoundments. A report of this symposium was published by OECD in 1970 (OECD, 1970). The Water Management Research Group became the Water Management Sector Group (WMSG) after formation of the Environment Committee in 1970 (OECD, 1975).

In 1971, after the formation of the Environment Committee, the WMSG established a Steering Group on Eutrophication Control. In 1973 and 1974, this group produced a series of reports concerning the effects of detergents, fertilizers and agricultural wastes, and phosphorus and nitrogen wastewater treatment processes



(From OECD, 1973a)

Figure 2. Organizational Structure of OECD Environment Committee.

on water quality. It also produced the Report of the Water Management Sector Group on Eutrophication in 1974. More significant, however, was the 1973 report entitled "Summary Report of the Agreed Monitoring Projects on Eutrophication of Waters" (OECD, 1973b). This report was prepared by a WMSG planning group on measurement and monitoring. It is this report which outlines the working plan for the international cooperative eutrophication study undertaken by OECD. The OECD North American Project is part of this cooperative effort.

OECD INTERNATIONAL COOPERATIVE PROGRAM FOR MONITORING OF INLAND WATERS

Objectives of Study

In order to better quantitatively define the eutrophication process and the factors which cause and control it, upon recommendation of the above-mentioned planning group, the WMSG established a program among the OECD member nations of measurement and monitoring of inland waters. This international effort was to coordinate measurements of nutrient budgets, chemical balances and biological productivity in water bodies in order to define guidelines for the selection of eutrophication control measures.

The objectives of the program were to refine the current knowledge concerning nutrient loadings and water body response, especially biological productivity, of selected water bodies so that guidelines could be established for predicting changes in trophic responses as a result of remedial treatments. The program was also to establish guidelines for predicting the reductions in nutrient loadings necessary to improve water quality in these water bodies. The ultimate goal was to economically assess the effects of eutrophication and introduce the control measures necessary to alleviate them (OECD, 1975). The specific objectives were:

1. promotion of an agreed common system of response parameters and analytical and sampling methods to allow comparison of eutrophication data between water bodies;

2. application of this common measuring system to selected categories of water bodies for a predetermined period, with the objective of obtaining a better understanding of the causes of eutrophication and the influence of nutrient loading on trophic status; and

3. promotion of a systematic exchange of information and experience on eutrophication and eutrophication control (OECD, 1973b; 1975).

Common Measurement System

Previous attempts to quantitatively categorize freshwater bodies in terms of tolerance to nutrient inputs, as manifested in their biological productivity, nutrient budgets and trophic levels, have been difficult because of the lack of comparable data for interrelating water bodies. Such lack of comparable data has greatly hindered development of criteria for predicting changes in water quality resulting from changes in nutrient loadings.

Consequently a common system of measurements was established early in the study. In addition to aiding in the choice of eutrophication control measures in a water body, the common system will also permit measurement of the effectiveness of a given control measure and the response of the water body to changing hydrological conditions.

The system of measurements recommended was divided into three categories: physical, chemical and biological. These categories were, in turn, divided into "essential" and "desirable" measurements. In addition, guidelines were established for the range of background data considered necessary for providing adequate geographical, morphometric, hydrological and ecological descriptions of a given water body.

The essential parameters were those considered necessary for establishing an accurate representation of trophic conditions in a given water body. These parameters would also allow a comparison of eutrophication data between water bodies. In addition, they would allow the assessment of the effectiveness of control measures initiated in an attempt to alleviate eutrophication problems.

Those parameters which were appropriate for large capacity laboratories or certain specialized laboratories were considered "desirable". In general, the desirable parameters were used to supplement the "essential" data (OECD, 1973b). A summary of these essential and desirable parameters is given in Table 2.

Recommended analytical methods were taken from FWPCA (1969), APHA _et al_. (1971) and Golterman (1971). Recommendations on sampling techniques included locations, depths and frequencies of sampling (OECD, 1973b).

Regional Approach

Recognizing that geographical, ecological, geological and morphometric factors are of major importance in the eutrophication process, the WMSG chose to employ a regional approach. Consequently the WMSG established four voluntary regional projects, each embracing a family of specified types of water bodies.

Eighteen member nations agreed to participate in these projects. There were three regionally-based projects and one

Table 2. SUMMARY OF ESSENTIAL AND DESIRABLE PARAMETERS IN OECD EUTROPHICATION STUDY

Category	Parameters
Physical	
Essential	Temperature, electrical conductivity, light penetration, color, total solar radiation.
Desirable	Turbidity.
Chemical	
Essential	pH, dissolved oxygen, phosphorus, nitrogen, silica, alkalinity, acidity, calcium, magnesium, sodium, potassium, sulfate, chloride, total iron.
Desirable	Other trace elements and other micro-pollutants (e.g. pesticides), hydrogen sulfide.
Biological	
Essential	Phytoplankton (chlorophyll a) primary productivity, organic carbon.
Desirable	Phytoplankton identification (by genera and counting); ^{14}C uptake, zooplankton identification (by genera and counting).

(From OECD, 1973b)

functionally-based project in the overall eutrophication study (OECD, 1973b). The regional organization and participating countries are illustrated in Figure 3. The coordination centers were to coordinate the activities within a given project and to act as vehicles of exchange of information between the four projects. Each individual project's groups of laboratories, assisted by its coordination center, was responsible for designing and establishing the necessary measurement procedures and data evaluation methods (OECD, 1973b).

Each project had a coordinator who was a senior scientist from one of the institutions or laboratories involved. Initially, the Coordinating Group was established as a link between the Technical Bureau and the WMSG. However, it was demonstrated that the Technical Bureau could adequately perform both the technical and managerial roles (OECD, 1975). The overall assessment and coordination of the four projects was the responsibility of a group of nationally nominated delegates from those countries participating in the study. This group was to synthesize the reports of the four projects into an optimal eutrophication control strategy and report to the WMSG, in principle once a year.

The program was planned to run four years, from the beginning of 1973 to the end of 1976. An overall analysis of the study is planned for 1977. Upon completion of the four-year period of measurements and study, it is expected that a symposium on the overall interpretation of the results will be convened in order to establish the extent to which nutrient loadings determine the rate of development of eutrophication (OECD, 1973b; 1975).

The four regional projects are characterized as follows:

1. Nordic Project - Reasonably comparable conditions exist in this project. These include the cool climate zone of the Baltic and North Sea areas; lakes resulting from retreat of the great Quaternary glaciers; comparable ecological conditions and equivalent level of economic development and pollution, and close political, cultural and scientific links.

2. Alpine Project - The Alpine regions are the source of many European waters. The Alpine waters are of great social and economic significance because they represent a great natural amenity and a source of considerable tourism. Their ecology is characterized by an abundant variety of species which are vulnerable to man's interventions. The Alpine zones represent similar hydrological conditions due to comparable geography, geology and ecology. The Alpine zones share certain river basins and commissions.

3. Reservoir and Shallow Lakes Project - This project includes man-made lakes and reservoirs and other

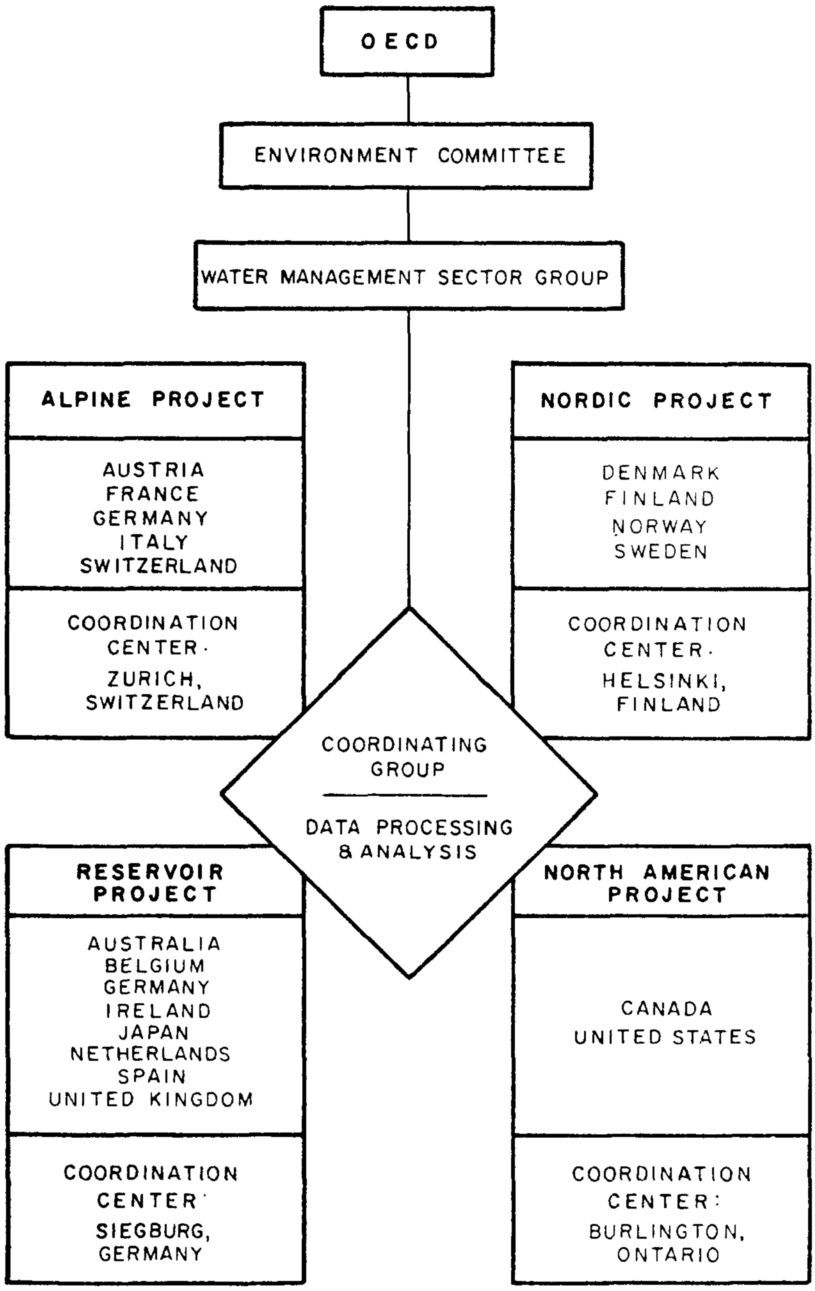


Figure 3. Organizational Outline of OECD Eutrophication Study.

comparable water bodies (i.e., shallow lakes, lagoons and estuarine waters). All are relatively shallow and have great economic and social values (e.g., water supply reserves, water sports, fishing, navigation, etc.). Water quality control by manipulation of hydrological or other factors is more feasible for these water bodies than for larger water bodies.

4. North American Project - In contrast to the other projects, this project is not restricted to studying specific types of water bodies. Rather, the trophic states of the involved water bodies span the trophic spectrum from oligotrophic to eutrophic (OECD, 1973b).

SECTION V

US OECD EUTROPHICATION STUDY

The major goal of the North American Project is similar to that of the other projects; namely, to determine the quantitative relationship between the nutrient loading and the resultant trophic state (i.e., degree of fertility) of a given water body.

Its specific objectives are as follows:

1. develop detailed nutrient budgets (phosphorus and nitrogen) for a selected group of water bodies;

2. assess the physical, chemical and biological characteristics of these selected water bodies;

3. relate the trophic states of the water bodies to their nutrient budgets and to their limnological and environmental characteristics; and

4. synthesize an optimal strategy, based on data from all four projects, for controlling eutrophication.

The North American Project consists of studying thirty-four water bodies in the United States and a larger number of water bodies in Canada. The director of the North American Project is R. Vollenweider of the Canada Centre for Inland Waters (CCIW) in Burlington, Ontario, Canada. The United States Environmental Protection Agency (US EPA) is the lead organization for the US portion of the North American Project. The US OECD study directors were N. Jaworski and J. Gakstatter (US EPA, 1973b). The 34 water bodies in the US OECD study are presented in Table 3 and their locations are illustrated in Figure 4.

The water bodies in the US OECD study differ considerably in their limnological characteristics and trophic states. It is the responsibility of the principal investigator for each water body to conduct the necessary measurements and to prepare the necessary reports for his water body. Nearly all of the water bodies selected for the US OECD study have been studied extensively in the past. Because of these factors and a lack of funds, no new sampling programs were initiated in the US OECD study. Some of the water bodies were also included in the US EPA's National Eutrophication Survey (NES), thereby providing a link between the US OECD studies and the NES studies.

Table 3. LIST OF WATER BODIES IN OECD NORTH AMERICAN PROJECT (US PORTION)

Water Body	Location	Tropic Status	Principal Investigator
Blackhawk, Camelot-Sherwood, Cox Hollow, Dutch Hollow, Redstone, Stewart, Twin Valley and Virginia	Wisconsin	Eutrophic	G. Fred Lee, Center for Environmental Studies, Univ. Texas at Dallas
Brownie, Calhoun, Cedar, Harriet and Isles	Minnesota	Eutrophic	J. Shapiro, Limnology Research Center, Univ. Minnesota
Canadarago	New York	Eutrophic	L. Hetling, Dept. Env. Conserv., State of New York
Cayuga	New York	Mesotrophic	R. Oglesby, Cornell Univ.
Dogfish, Lamb and Meander	Minnesota	Oligotrophic	S. Tarapchak, NOAA Great Lakes Env. Res. Lab, Ann Arbor, Mich.
George	New York	Oligotrophic-Mesotrophic	N. Clesceri, Rensselaer Polytechnic Inst.
Kerr Reservoir	N. Carolina, Virginia	Eutrophic-Mesotrophic	C. Weiss, Univ. North Carolina.
Mendota	Wisconsin	Eutrophic (Changing)	G. Fred Lee, Center for Environmental Studies, Univ. Texas at Dallas

Table 3 (continued). LIST OF WATER BODIES IN OECD NORTH AMERICAN PROJECT (US PORTION)

Water Body	Location	Trophic Status	Principal Investigator
Michigan Open waters	Wisconsin, Michigan, Illinois & Indiana	Oligotrophic	G. Fred Lee, Center for Environmental Studies, Univ. Texas at Dallas *and*
Nearshore Waters		Mesotrophic	C. Schelske, Great Lakes Research Division, Univ. Michigan
Minnetonka	Minnesota	Eutrophic (Changing)	R. Megard, Limnology Research Center, Univ. Minnesota
Potomac Estuary	Maryland, Virginia	Ultra-Eutrophic	N. Jaworski, US EPA, Corvallis, Oregon
Sallie	Minnesota	Eutrophic	J. Neel, Univ. North Dakota
Sammamish	Washington	Mesotrophic	E. Welch, Univ. Washington
Shagawa	Minnesota	Eutrophic	K. Malueg, US EPA, Corvallis, Oregon
Tahoe	California, Nevada	Ultra-Oligotrophic	C. Goldman, Univ. California at Davis
Twin Lakes	Ohio	Eutrophic (Changing)	D. Cooke, Kent State Univ.
Waldo	Oregon	Ultra-Oligotrophic	C. Powers, US EPA, Corvallis, Oregon

Table 3 (continued). LIST OF WATER BODIES IN OECD NORTH AMERICAN PROJECT (US PORTION)

Water Body	Location	Trophic Status	Principal Investigator
Washington	Washington	Meostrophic	W.T. Edmondson, Univ. Washington
Weir	Florida	Mesotrophic	P. Brezonik, Univ. Florida
Wingra	Wisconsin	Eutrophic	G. Fred Lee, Center for Environmental Studies, Univ. Texas at Dallas
Trophic Status Index Study			J. Shapiro, Limnology Research Center, Univ. Minnesota
Summarization Report - US OECD Project			G. Fred Lee and W. Rast, Center for Environmental Studies, Univ. Texas at Dallas

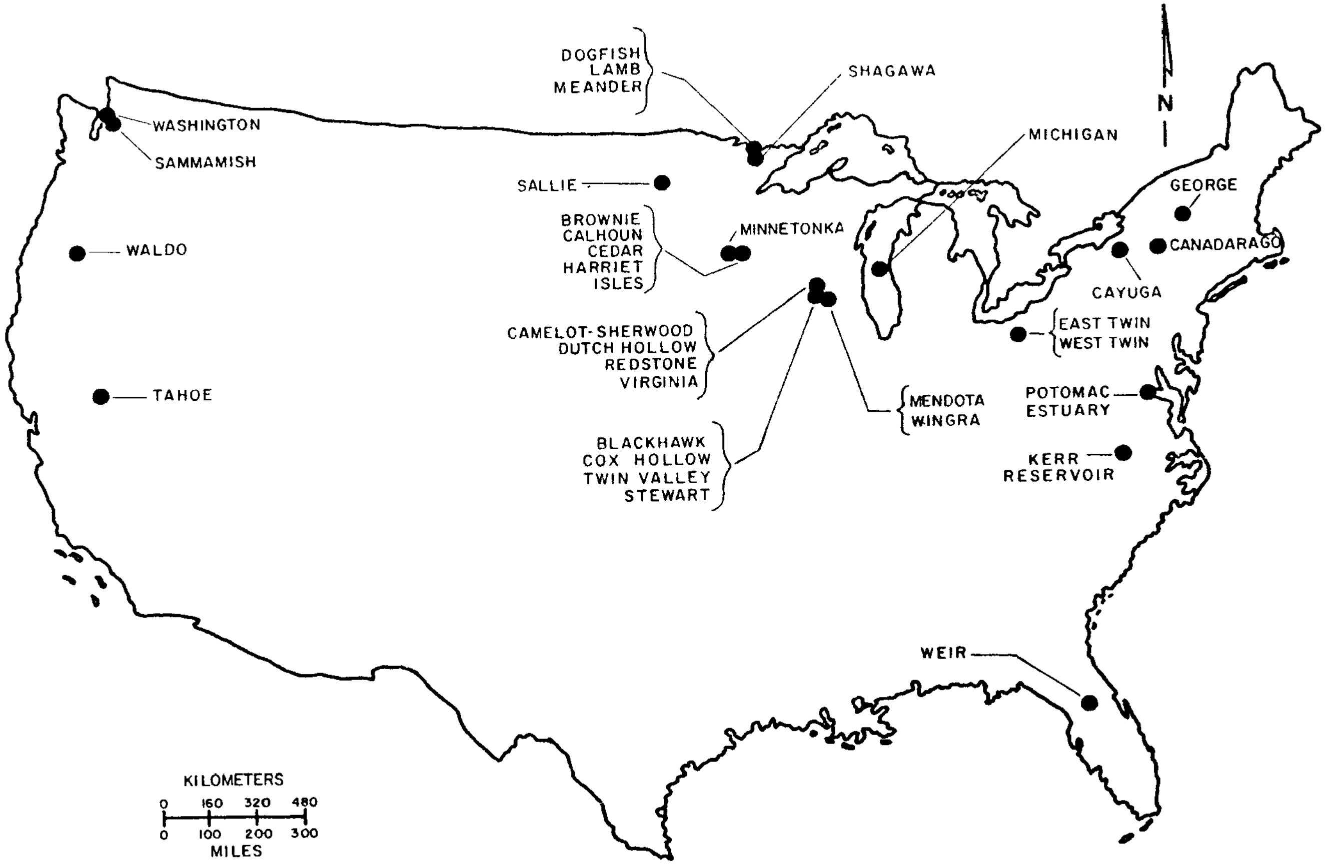


Figure 4. Locations of US OECD Water Bodies.

GENERAL CHARACTERISTICS OF US OECD WATER BODIES

The general characteristics of the water bodies in the US OECD study are presented in Table 4, which indicates that the 34 water bodies of the US OECD study include 24 lakes, nine impoundments and one estuary. Thus, 71 percent of the water bodies in the US OECD study are lakes and 26 percent are impoundments. However, several of these water bodies are divided into separate arms or regions (e.g., Kerr Reservoir and the Potomac Estuary). When these separate regions are considered, there are 38 US water bodies in the US OECD eutrophication study. Furthermore, several of the US OECD water bodies have been previously examined and have subsequently undergone remedial treatment for eutrophication (e.g., Minnetonka, Twin Lakes, Washington). Thus, although 38 water bodies are included in the US OECD study, a total of 47 individual nutrient loading-trophic response relationships were examined.

The principal investigators classified 24 of the water bodies as eutrophic (63 percent), seven as mesotrophic (18 percent) and seven as oligotrophic (18 percent) at the time of the US OECD study. These percentages reflect the investigator-indicated trophic states at the time of submission of final reports.

Twenty-eight (74 percent) of the water bodies have mean depths less than ten meters while ten (26 percent) have mean depths greater than ten meters. The mean depths range from 1.7 meters (Lake Virginia) to 313 meters (Lake Tahoe). The watershed areas range from 47 hectares (Brownie Lake) to 1.76×10^7 hectares (Lake Michigan). Sixteen (42 percent) of the water bodies have surface areas greater than 1000 hectares. Twenty-three (61 percent) of the water bodies have hydraulic residence times (i.e., water body volume/annual inflow volume) of greater than one year. The hydraulic residence times range from 0.08 yr (Lake Stewart) to 700 yr (Lake Tahoe). Twenty-four (63 percent) have mean specific conductances of 200 μmhos/cm (25°C) or greater.

Of the 24 water bodies with mean specific conductances greater than 200 μmhos/cm, 21 were classified eutrophic, two mesotrophic and one oligotrophic. As expected, the single estuary studied had the highest mean specific conductance, ranging from 200-500 μmhos/cm (25°C) at the fresh water input to 26,000 μmhos/cm at the saline end of the estuary.

Of the 13 water bodies with less than 200 μmhos/cm mean specific conductance, seven were oligotrophic, four mesotrophic, and two eutrophic. Ultra-oligotrophic Lake Waldo exhibited the lowest reading, 3 μmhos/cm (25°C).

The mean alkalinities ranged from 2 mg/l as $CaCO_3$ (Lake Waldo) to 248 mg/l (Canadarago Lake). The distribution was relatively even, with 18 (47 percent) having mean alkalinities greater than 100 mg/l as $CaCO_3$.

Table 4. CHARACTERISTICS OF US OECD WATER BODIES[a]

WATER BODY (location)	Trophic Status[b]	Water Body Type[c]	Watershed Area[d] ($\times 10^6$ m^2)	Water Body Surface Area ($\times 10^5$ m^2)	Mean Depth[e] (m)	Mean Hydraulic Residence Time[f] (yr)	Mean Secchi Depth (m)	Mean Conductivity (μmhos/cm @ 25°C)	Mean Alkalinity (mg/l as $CaCO_3$)
LAKE BLACKHAWK (Wisc.)	E	I	36.3	8.90	4.9	0.5	3.6	471	227
BROWNIE LAKE (Minn.)	E	L	0.47	0.73	6.8	2.0	1.5	400-475	123-136
LAKE CALHOUN (Minn.)	E	L	7.61	17.0	10.6	3.6	2.1	400-500	80-123
CAMELOT-SHERWOOD COMPLEX (Wisc.)	E	I	90.6	28.3	3.0	0.09-0.14	2.0	311	125
CANADARAGO LAKE (N.Y.)	E	L	182	75.9	7.7	0.6	1.8	223	248
CAYUGA LAKE (N.Y.)	M	L	2030	1720	54.5	8.6	2.3	575	102
CEDAR LAKE (Minn.)	E	L	1.63	6.90	6.1	3.3	1.8	400	71-109
COX HOLLOW LAKE (Wisc.)	E	I	16.1	3.88	3.8	0.5-0.7	1.5	440	205
DOGFISH LAKE (Minn.)	O	L	0.88	2.91	4.0	3.5	2.5-2.7	16-17	8-10
DUTCH HOLLOW LAKE (Wisc.)	E	I	12.5	8.50	3.0	1.8	0.8	252	133

Table 4 (continued). CHARACTERISTICS OF US OECD WATER BODIES[a]

WATER BODY (Location)	Trophic Status[b]	Water Body Type[c]	Watershed Area[d] ($x10^6$ m^2)	Water Body Surface Area ($x10^5$ m^2)	Mean Depth[e] (m)	Mean Hydraulic Residence Time[f] (yr)	Mean Secchi Depth (m)	Mean Conductivity (µmhos/cm @ 25°C)	Mean Alkalinity (mg/l as $CaCO_3$)
LAKE GEORGE	O-M	L	606	1140	18.0	8.0	8.5	86	21
LAKE HARRIET (Minn.)	E	L	4.80	14.3	8.8	2.4	2.4	360-425	92-124
LAKE OF THE ISLES (Minn.)	E	L	2.85	4.20	2.7	0.6	1.0	380-470	68-131
KERR RESERVOIR (N.Carolina-Vir.)	E-M	I	20,200	1754	-	-	-	-	-
Roanoke Arm		-	-	1250	10.3	0.2	1.4	100	28
Nutbush Arm		-	-	504	8.2	5.1	1.2	123	22
LAMB LAKE (Minn.)	O	L	1.96	3.97	4.0	2.3	1.8-2.2	47	30-36
MEANDER LAKE (Minn.)	O	L	1.69	3.60	5.0	2.7	3.0-3.1	17-20	8
LAKE MENDOTA (Wisc.)	E	L	686	394	12.0	4.5	3.0	300	160
LAKE MICHIGAN (Wisc., Mich., Ill., Ind.)	O-M	L	176,000	580,000	84	30-100[h]	-	-	-

Table 4 (continued). CHARACTERISTICS OF US OECD WATER BODIES[a]

WATER BODY (Location)	Trophic Status[b]	Water Body Type[c]	Watershed Area[d] ($x10^6$ m^2)	Water Body Surface Area ($x10^5$ m^2)	Mean Depth[e] (m)	Mean Hydraulic Residence Time[f] (yr)	Mean Secchi Depth (m)	Mean Conductivity (µmhos/cm @ 25°C)	Mean Alkalinity (mg/l as $CaCO_3$)
LAKE MICHIGAN (cont'd)									
Nearshore Waters	M	-	-	-	-	-	2.3	265	107
Offshore Waters	M	-	-	-	-	-	7.0	260	106
Open Lake Waters	O	-	-	-	-	-	-	255	113
LAKE MINNETONKA (Minn.)		L	371[g]	262	8.3	6.3[g]	-	-	-
Pre-sewage Treatment(1969)	E	L	371[g]	262	8.3	-	1.5	317	250
Post-sewage Treatment(1973)	E→M	L	371[g]	262	8.3	-	1.8	-	250
POTOMAC ESTUARY (Maryland, Vir.)	U-E	E	38,000	9644					
Upper Reach	-	-	-	574	4.8	0.04	0.4-0.8	200-500	70-110
Middle Reach	-	-	-	2120	5.1	0.18	0.5-1.3	600-17,000	60- 85
Lower Reach	-	-	-	6950	7.2	0.85	1.0-2.3	17,000-26,000	65- 85
LAKE REDSTONE (Wisc.)	E	I	76.7	25.2	4.3	0.7-1.0	1.6	260	125

Table 4 (continued). CHARACTERISTICS OF US OECD WATER BODIES[a]

WATER BODY (Location)	Trophic Status[b]	Water Body Type[c]	Watershed Area[d] ($x10^6$ m^2)	Water Body Surface Area ($x10^5$ m^2)	Mean Depth[e] (m)	Mean Hydraulic Residence Time[f] (yr)	Mean Secchi Depth (m)	Mean Conductivity (µmhos/cm @ 25°C)	Mean Alkalinity (mg/l as $CaCO_3$)
LAKE SALLIE (Minn.)	E	L	1540	53.0	6.4	1.1-1.8	-	280-360	162
LAKE SAMMAMISH (Wash.)	M	L	273	198	17.7	1.8	3.3	94	33
SHAGAWA LAKE (Minn.)	E	L	269	92.0	5.7	0.8	2.3	60	22
LAKE STEWART (Wisc.)	E	I	2.07	0.25	1.9	0.08	1.4	540	213
LAKE TAHOE (Calif.,Nev.)	U-O	L	1310	4990	313	700	28	92	43
TWIN LAKES (Ohio)	-	L	3.34	-	-	-	-	-	-
EAST TWIN LAKE	-	L	-	2.69	5.0	-	-	-	-
Pre-sewage Treatment(1972)	E	L	-	2.69	5.0	0.80	1.6	374	-
Post-sewage Treatment(1974)	E	L	-	2.69	5.0	0.50	1.9	366	105
WEST TWIN LAKE	-	L	-	3.40	4.3	-	-	-	-
Pre-sewage Treatment(1972)	E	L	-	3.40	4.3	1.6	2.2	411	-
Post-sewage Treatment(1974)	E	L	-	3.40	4.3	1.0	2.3	380	106

Table 4 (continued). CHARACTERISTICS OF US OECD WATER BODIES[a]

WATER BODY (Location)	Trophic Status[b]	Water Body Type[c]	Watershed Area[d] ($\times 10^6$ m^2)	Water Body Surface Area ($\times 10^5$ m^2)	Mean Depth[e] (m)	Mean Hydraulic Residence Time[f] (yr)	Mean Secchi Depth (m)	Mean Conductivity (µmhos/cm @ 25°C)	Mean Alkalinity (mg/l as $CaCO_3$)
TWIN VALLEY LAKE (Wisc.)	E	I	31.1	6.07	3.8	0.4-0.5	1.5	370	175
LAKE VIRGINIA (Wisc.)	E	I	6.48	1.82	1.7	0.9-2.8	1.2	230	64
WALDO LAKE (Ore.)	U-O	L	79	270	36	21	28	3	2
LAKE WASHINGTON (Wash.)	-	L	1590	876	33	2.4	-	-	-
Pre-sewage Diversion (1964)	E	L	1590	876	33	2.4	1.2	80	25
Post-sewage Diversion (1974)	M	L	1590	876	33	2.4	3.8	81	45
LAKE WEIR (Fla.)	M	L	46.0	240	6.3	4.2	1.9	133	12
LAKE WINGRA (Wisc.)	E	L	14.0	13.7	2.4	0.4	1.3	-	153

[a]As reported by US OECD investigators. See Summary Sheets (Appendix II)

[b]Investigator-indicated trophic status: (U-E) = Ultra-Eutrophic
(E) = Eutrophic
(M) = Mesotrophic
(O) = Oligotrophic
(U-O) = Ultra-Oligotrophic

Table 4 (continued). CHARACTERISTICS OF US OECD WATER BODIES[a]

EXPLANATION: (continued)

[c]Water body type: E = Estuary
I = Impoundment
L = Lake

[d]Includes lake surface area

[e]Mean depth = water body volume (m^3)/water body surface area (m^2)

[f]Hydraulic residence time = water body volume (m^3)/annual inflow volume (m^3/yr)

[g]*Values for whole lake. All other data is only for Lower Lake Minnetonka*

[h]Range of values as reported in the literature; most accurate range is assumed to be 70-100 years. See Piwoni et al. (1976) for discussion of Lake Michigan hydraulic residence time.

Dash (-) indicates data not available.

Twenty-eight (74 percent) had mean Secchi depths less than three meters. No Secchi data were available for two water bodies. Of the 28 water bodies with Secchi depths less than three meters, 22 were classified by their respective investigators as eutrophic, five mesotrophic and one oligotrophic (Dogfish Lake). Within the eight water bodies of three meters or greater Secchi depths, five were classified oligotrophic, one mesotrophic and two eutrophic (Lakes Blackhawk and Mendota). The mean Secchi depths ranged from 0.6 meters in the Upper Reach of the Potomac Estuary to greater than 28 meters (Lakes Tahoe and Waldo).

DATA REPORTING METHODOLOGY

The general approach involved in the US OECD study is presented in the Final Report Outline (Appendix I). This Final Report Outline was prepared by the North American Project participants and served both as a guide to the types of information and studies needed in the North American Project and as an outline for the presentation of the data generated in the North American Project in standardized Final Reports. Part of the information in the Final Report Outline was suggested by the WMSG as necessary "background data" (OECD, 1973b).

The Final Report Outline begins with a short introductory section, followed by a brief geographical description of the water body. This includes its latitude, longitude and altitude, the watershed area, general climate data, general geological description, vegetation, watershed population, land usage and wastewater discharges into the water body. Next is a brief morphometric and hydrologic description of the water body, including its surface area, volume, mean and maximum depths, ratio of epilimnion to hypolimnion, duration of stratification, lake sediment types, seasonal precipitation variation, water budget, water currents and hydraulic residence time. This is followed by a limnological characterization of the water body, including a physical, chemical and biological summary. A nutrient budget summary, including phosphorus and nitrogen inputs, follows the limnological characterization. Finally, there is a discussion section which includes a delineation of water body trophic status and discussion of the general limnological characteristics. In addition, the degree of correlation between the water body nutrient loading and trophic response is discussed in detail. These two parameters are also to be discussed in relation to the water body's general limnological characteristics.

The US OECD study "Summary Sheets" (Appendix II) were devised to summarize the important loading and response parameters of the US OECD water bodies. They include the water body name and type, watershed and water body surface area, mean depth, water residence time, important trophic response parameters (e.g., nutrient and chlorophyll a concentrations, primary productivity) and nutrient loadings. The Summary Sheets and the Final Report Outline were prepared to allow the presentation of data in a standardized form.

US OECD EUTROPHICATION STUDY AND OTHER US EUTROPHICATION CONTROL PROGRAMS

National Eutrophication Survey

Several years ago, the US EPA (1975a) initiated the National Eutrophication Survey. This Survey was designed to study approximately 800 water bodies throughout the US for which estimated nutrient load-response relationships would be ascertained. Because of funding limitations, sampling of tributaries and water bodies was limited to one year and was not intensive. The US OECD eutrophication study provides similar information for a smaller number of water bodies and was generally based on a much more intense sampling program. For the water bodies common to both programs, a comparison of the two approaches will aid the US EPA and other water pollution regulatory agencies in assessing the validity of the results and conclusions from the National Eutrophication Survey.

Public Law 92-500

Section 314-A of Public Law 92-500 requires all the states in the US to classify their publicly-owned water bodies as to trophic status. It further requires the states to initiate eutrophication control measures in water bodies deemed excessively fertile. Thus, the overall aims of the US OECD eutrophication study, the US EPA's NES study and the intent of Public Law 92-500, Section 314-A, are generally identical. They are to ascertain what trophic classification or index system should be used, what parameters should be measured, how a given set of conditions in a water body can be related to its trophic status, how one predicts response of a water body to a change in a chemical, biological or physical parameter and what the aquatic plant trophic response will be to a given water body's nutrient input. By attempting to answer questions of this type, the US OECD eutrophication study can be used by the states to help them fulfill the mandate of Section 314-A of Public Law 92-500.

Public Law 92-500 also requires the US EPA to develop water quality criteria. In October, 1973 the US EPA released draft proposed criteria for public comment (US EPA, 1973c). In November, 1975 the US EPA released revised draft Quality Criteria for Water (US EPA, 1975b) and again asked for comment. While no criteria were proposed for phosphorus as an aquatic plant nutrient, the US EPA suggested in the November 1975 criteria that a nutrient loading-response relationship similar to those being investigated in the US OECD eutrophication study be adopted.

USE OF N:P RATIOS IN DETERMINING THE AQUATIC PLANT GROWTH LIMITING NUTRIENT IN NATURAL WATERS

The role of phosphorus and nitrogen as aquatic plant (i.e., algae and macrophytes) nutrients in the primary productivity and, hence, in the eutrophication of natural waters has been well-documented (Sawyer, 1947; American Water Works Association, 1966; Vollenweider, 1968; Edmondson, 1970b; Lee, 1971; Ryther and Dunstan, 1971; Maloney *et al.*, 1972; Powers *et al.*, 1972; Martin and Goff, 1972; Shannon and Brezonik, 1972; Brezonik, 1973; Lee, 1973; Vallentyne, 1974; United States Environmental Protection Agency, 1974a; Schindler and Fee, 1974; Vollenweider, 1975a; and Jones and Bachmann, 1975, to cite but a few). The effects of man-induced nutrient inputs, as opposed to natural nutrient inputs, in accelerating the eutrophication process has also been studied in detail (Sawyer, 1952; Curry and Wilson, 1955; Shapiro and Ribeiro, 1965; Maloney, 1966; Vollenweider, 1968; Bartsch, 1970; Stumm and Morgan, 1970; Bartsch, 1972; Edmondson, 1972; Beeton and Edmondson, 1972; and Vallentyne, 1974). Various other elements or compounds have been suggested as affecting or limiting the eutrophication process, including iron, molybdenum, nitrate and sulfate, vitamins and other organic growth factors, carbon and silicon (Goldman, 1960; Menzel and Ryther, 1961; Goldman and Wetzel, 1963; Goldman, 1964; Lange, 1967; Kuentzel, 1969; Provasoli, 1969; Kerr *et al.*, 1970; Schelske and Stoermer, 1972). However, most of these effects are either site-specific, or else are temporal in nature and do not persist over the annual cycle. Today, it is generally accepted that the phosphorus and nitrogen in a water body, rather than the above-mentioned compounds, control or limit the eutrophication process through their roles as aquatic plant nutrients in the primary productivity of the water body. However, not only are the absolute quantities of phosphorus and nitrogen in a water body of importance in the eutrophication process, but also their relative quantities seem to be a key factor in determining which of these two elements will limit the overall process.

The Limiting Nutrient Concept

A nutrient will be consumed or assimilated by an organism in proportion to the organism's need for that nutrient. However, it was noted as early as 1840 by Justus Liebig that growth of a crop was not generally limited by the nutrients needed in large quantities, which were often abundant in the environment, but rather by the nutrients needed in minute quantities, which were often scarce. This observation forms the basis of one of the oldest laws of plant nutrition, Liebig's "Law of the Minimum" (Odum, 1971). Simply stated, Liebig's law states that growth of an organism is limited by the substance or foodstuff which is available to it in the minimal quantity relative to its needs for growth or reproduction. This principle has also been applied to factors other than nutrients, including light and temperature.

However, for the purposes of this discussion, the limiting nutrient concept, as Liebig's Law of the Minimum has come to be called, will be restricted to aquatic plant nutrients.

Nitrogen and Phosphorus as Limiting Nutrients

The nutrients (i.e., elements or compounds) needed in relatively large quantities by aquatic plants include carbon, hydrogen, oxygen, sulfur, potassium, calcium, magnesium, nitrogen and phosphorus (Fruh, 1967). In addition, there is a requirement for traces of micronutrients as listed in Table 5.

TABLE 5. SUMMARY OF AQUATIC PLANT MICRONUTRIENT REQUIREMENTS

Process	Trace Element Required
Photosynthesis	Manganese, iron, chloride, zinc and vanadium
Nitrogen Fixation	Iron, boron, molybdenum and cobalt
Other Functions	Manganese, boron, cobalt, copper and silicon

(After Shannon, 1965, as cited in Fruh, 1967)

Among these macro- and micronutrient requirements, nitrogen and phosphorus are generally considered to be the aquatic plant nutrients of major importance in the eutrophication process.

Recently, the possible role of carbon as a limiting nutrient has been proposed (Lange, 1967; Kuentzel, 1969; Kerr *et al.*, 1970). However, the work underlying the so-called "Lange-Kuentzel-Kerr thesis" has been questioned on several grounds (Shapiro, 1970; Schindler, 1971; 1977; Fuhs *et al.*, 1972; Goldman *et al.*, 1972). Goldman *et al.* (1972) have reported that the results of Kerr *et al.* (1970), indicating CO_2 to be the limiting nutrient in their experiments, were due primarily to faulty experimental design. The conclusions of Kerr *et al.* (1970) were supported mainly by laboratory data with samples which contained surplus phosphorus and a limited CO_2 content. Consequently, carbon was limiting almost from the beginning of their experiments. A similar situation is frequently seen in wastewater stabilization ponds where, because of the excessive quantities of phosphorus and nitrogen relative to carbon, total algal productivity is known to be limited by carbon (Goldman *et al.*, 1972). Such a situation generally does not appear to occur in natural waters. Maloney *et al.* (1972), in laboratory assays on water from nine Oregon lakes, and Powers et al. (1972), in field experiments on lakes in Oregon and Minnesota, demonstrated that carbon addition to the waters had no effect on algal growth rates. Further, there appeared to be no correlation between algal rates and carbon concentration in the water bodies. Schindler (1977) reported that the bottle bioassay experiments used to test the carbon limitation theory were faulty in that they eliminated the turbulence of water and its interaction with the overlying atmosphere and because no attempt was made in the experiments to simulate the proportion of alkalinity supplied by hydroxyl ions in natural waters which affects the rate at which carbon is taken into the aquatic ecosystem.

Shapiro (1973) has demonstrated that a shift from blue-green algae to green algae resulted when CO_2 was added to their water. Presumably, a shift from green algae to blue-green algae would occur in natural waters as the CO_2 content of the water was depleted. Shapiro concluded that this shift to blue-green algae would likely occur because they appear to be more efficient in utilizing CO_2 in waters of low CO_2 content. This shift in algal types, rather than a general reduction in algal biomass, implies that the total algal content remains relatively unaffected in waters low in CO_2. Rather, there is a shift to blue-green algal types because of their nutrient uptake kinetics in low CO_2 waters. Thus, a low CO_2 content in natural waters will not necessarily limit algal growth, but rather can shift the dominant algal types from green to blue-green algae without significantly affecting the overall primary productivity and algal biomass.

Recently James and Lee (1974) have shown similar results in examination of inorganic carbon limitation in natural waters. According to their model, inorganic carbon limitation could conceivably occur in low alkalinity waters. However, they also indicate that the types, rather than quantities, of algae present in a water body could be significantly affected by the amounts and forms of inorganic carbon present. Under such conditions, there may be no noticeable change in total algal biomass, even though the inorganic carbon content of the water may drop to apparently growth-limiting levels.

As a result of these above-mentioned studies, it is generally accepted today among investigators that carbon will not usually be a limiting nutrient in natural waters, except under certain well-defined conditions. These special conditions would include sewage lagoons, already eutrophic water bodies, laboratory flasks with artificial media or special situations affecting the amounts of available inorganic carbon, such as very low alkalinity lakes or extremely hard water bodies (Goldman et al., 1972; James and Lee, 1974). As such conditions occur infrequently in nature, carbon limitation of total algal growth would be rare in most natural waters.

In addition to the many works reported on the role of nitrogen and phosphorus in the eutrophication of natural waters (Sawyer, 1947; Hutchinson, 1957; Vollenweider, 1968; Lee, 1971; Vallentyne, 1974; Vollenweider and Dillon, 1974), it has also been observed that these two nutrients are usually present only in small quantities in natural waters during periods of excessive algal growths (Mackenthun et al., 1964, as cited in Fruh, 1967). Vallentyne (1974) has indicated the special significance of nitrogen and phosphorus among the 15 to 20 elements commonly needed for the growth of aquatic plants by calculating the demand:supply ratios of these essential elements. According to Vallentyne (1974), aquatic plants have a certain demand for nutrients, for their growth and reproduction, in proportion to the quantities of the nutrients in their cells. When one or more of these nutrients is present in short supply relative to the others, then the overall primary productivity of the aquatic plant population will be limited by the rates of supply of these nutrients. Thus, a "demand:supply" ratio can reveal the nutrient most likely to limit productivity. The higher this demand:supply ratio, the more a particular nutrient will limit growth. The demand:supply ratios, based on a "world average", were calculated by determination of the chemical composition of an average aquatic plant community and dividing this composition by the mean chemical composition of the river waters of the world. These demand:supply ratios are presented in Table 6. The dominant role of phosphorus and nitrogen is clearly illustrated in Table 6 by their very high demand: supply ratios, relative to all the other elements normally needed by aquatic plants. This is especially prominent during the midsummer (i.e., during the growing season).

TABLE 6. DEMAND:SUPPLY RATIOS FOR THE MAJOR AQUATIC PLANT NUTRIENTS

	Demand:Supply	
Element	Late Winter[a]	Midsummer[b]
Phosphorus	80,000	up to 800,000
Nitrogen	30,000	up to 300,000
Carbon	5,000	up to 6,000
Iron, Silicon	Variable, but generally low	
All other elements	< 1,000	

[a]Prior to spring bloom

[b]At algal maximum growth period

(Taken from Vallentyne, 1974)

Thus, nitrogen and phosphorus are the two elements most often found to be limiting aquatic plant growths. There have been a few instances in which other elements have been found to have a cause-effect role in limiting growth, including silicon (Schelske and Stoermer, 1972) and iron (Welch et al., 1975). However, the overall importance of these exceptions is not comparable to the dominant roles played by phosphorus and nitrogen in the eutrophication process.

Interaction Between Biotic and Abiotic Factors in Determining Limiting Nutrients and Algae Nutrient Stoichiometry

It is a long-recognized principle in ecology that interactions between organisms and their environment are reciprocal (Redfield, 1958; Odum, 1971). The environment determines the conditions under which an organism lives. Organisms respond to changes in their physical environment by altering their metabolism or growth requirements. Algae can directly influence their environment by changing the concentration of nutrients and other substances in the water by metabolic uptake, transformation, storage and release. This is usually related to reciprocal changes in algal biomass. This exchange between algal biomass and nutrient concentration in natural waters is a cyclic process, which must always be considered in any attempt to understand the chemistry in aquatic environments (Redfield et al., 1963; Stumm and Morgan, 1970).

This cyclical exchange is a two-phase process, including synthesis and regeneration. With algae, the synthesis phase consists of withdrawal of nutrients, especially nitrogen and phosphorus, from the water during photosynthesis. These nutrients are withdrawn from the water in the proportions required for growth of the algae. The regeneration phase occurs when the elements are returned to the water as decomposition products and excretions of the algae, the higher trophic level species which feed upon them and the microorganisms which decompose their organic debris (Redfield et al., 1963).

The proportions in which algal nutrients in natural waters enter into the cyclical process described above is determined by the elementary composition of the algal biomass. It is generally accepted that algae need a relatively fixed atomic ratio of carbon to nitrogen to phosphorus of 106 to 16 to 1 (i.e., (106C:16N:1P) (Redfield, 1958; Redfield et al., 1963; Vollenweider, 1968; Ketchum, 1969; Lee, 1973). This observation has a basis in the simple stoichiometry of the photosynthesis-respiration reaction as it occurs in nature, as illustrated in the following equation:

$$106\ CO_2 + 16\ NO_3^- + HPO_4^+ + 122\ H_2O + 18\ H^+ + \text{trace elements}$$

$$+ \text{energy} \underset{\text{respiration}}{\overset{\text{photosynthesis}}{\rightleftarrows}} \underset{\text{algal protoplasm}}{\{C_{106}H_{263}O_{110}N_{16}P_1\}} + 138\ O_2$$

(Taken from Stumm and Morgan, 1970)

The 106C:16N:1P atomic ratio was obtained from the early work of Redfield (1934) and Fleming (1940), as cited in Redfield *et al.* (1963), who examined the organic matter in plankton samples obtained in sea water for the relative quantities of the principal elements present in the plankton. The C:N:P atomic ratio values represent an average of the carbon, nitrogen and phosphorus content present in phytoplankton and zooplankton, as illustrated in Table 7.

TABLE 7. ATOMIC RATIOS OF C, N AND P PRESENT IN PLANKTON

	C	N	P
Zooplankton	103	16.5	1
Phytoplankton	108	15.5	1
Average Value	106	16	1

(Taken from Redfield *et al.*, 1963)

In this discussion, attention is centered on nitrogen and phosphorus since it is the relative quantities of these two elements, rather than carbon, that is likely to limit or control algal growth, and thereby the eutrophication process, presuming all other physical and chemical factors are optimal for algal growth.

The N:P ratios listed above may change as a function of the aquatic environment. Harris and Riley (1956, as cited in Redfield *et al.*, 1963), studying plankton from Long Island Sound, reported that while the average N:P atomic ratio in phytoplankton in their study was 16:1, the average zooplankton N:P ratio was 24:1. Further, differences during the annual cycle varied as much as 25 percent, with zooplankton having the highest N:P ratios in winter and spring. Ketchum and Redfield (1949, as cited in Redfield *et al.*, 1963), using mass cultures of the freshwater algae *Chlorella pyrenoidosa*, demonstrated that a wide variation in the N:P ratio can occur under extremes of nitrogen and phosphorus concentrations in the growth medium. In their experiments, normal algal culture cells contained an N:P ratio of about 6:1. By contrast, phosphorus deficient cells exhibited an N:P ratio as high as 31:1, while nitrogen deficient cells would show an N:P ratio of 3:1 or less.

Fuhs *et al.* (1972), using *Cyclotella nana* in laboratory cultures, have shown that under severe phosphorus limitation, the N:P ratio can rise to 35:1. It can drop to very low levels when nitrogen is limiting as a result of "luxury consumption" of phosphorus. Fitzgerald (1969) has also demonstrated, with the use of enzymatic and tissue assay procedures, that the N:P ratio in algae and aquatic weeds can vary widely, depending on whether nitrogen or phosphorus is present in excess in the growth medium.

However, while laboratory studies have demonstrated a marked variation in algal N:P ratios because of the relative quantities of nitrogen and phosphorus in the growth medium, field studies have shown that rarely do such variations occur in natural waters. Generally, neither phosphorus nor nitrogen are present in natural waters in excessive quantities relative to the other. Consequently, algae in natural waters do not usually contain nitrogen and phosphorus in the ratios induced by the artificial conditions of severe phosphorus or nitrogen limitation in the laboratory studies. This is illustrated in examination of the nitrogen and phosphorus content of algae from natural waters in the southeastern US (Table 8).

TABLE 8. CHEMICAL COMPOSITION OF SOME ALGAE FROM PONDS AND LAKES IN THE SOUTHEASTERN US

Algae	N:P Atomic Ratio
Chara	22:1
Pithophora	20:1
Spirogyra	33:1
Giant *Spirogyra*	22:1
Rhizoclonium	18:1
Oedogonium	73:1
Mougeotis	16:1
Anabaena	27:1
Cladophora	9:1
Euglena	27:1
Hydrodictyon	36:1
Microcystis	27:1
Lyngbya	36:1
Nitella	27:1
Amphizomenon	16:1

(Based on Federal Water Pollution Control Administration, 1968, as cited in Goldman *et al.*, 1972)

Examination of Table 8 shows, with few exceptions, that in general the N:P ratio of the algae varies between 16:1 to 27:1. This ratio is smaller than the 35:1 ratio shown with *Cyclotella nana* under severe phosphorus limitation in laboratory cultures (Fuhs *et al.*, 1972) and higher than that shown with *Chlorella pyrenoidosa* under severe nitrogen limitation (Ketchum and Redfield, 1949, as cited in Redfield *et al.*, 1963). If the minimum and maximum values are omitted, the mean N:P atomic ratio of the algae is 24:1 (standard deviation = 8). Even if all values are included, the mean N:P atomic ratio in Table 8 is 27:1 (standard deviation = 15). Thus, generally, algal populations in natural waters do not exhibit the extremes in cellular N:P ratios seen in algal laboratory cultures.

Thus, even in spite of some variation, it is generally accepted that the N:P atomic ratio in natural algal populations remains constant enough to be used in making reasonable predictions as to which of these two elements is likely to limit algal growths in natural waters.

The Limiting Nutrient Concept As Applied In The US OECD Eutrophication Study

Presumably, as a result of the photosynthesis reaction, algae will assimilate nitrogen and phosphorus from their aquatic environment in a stoichiometric atomic ratio of approximately 16N:1P until one of these two nutrients becomes depleted in the water body. At that time, the nutrient present in the water body in the lowest concentration, relative to the stoichiometric needs of the algae, will limit subsequent growth of the algae. An examination of the water body at that time for its content of nitrogen and phosphorus would indicate which of these nutrients had been depleted by the algae (i.e., which nutrient was the limiting nutrient). If the N:P atomic ratio in the water body fell below 16, this would mean there were less than 16 nitrogen atoms per each phosphorus atom in the water. Since this is below the 16N:1P stoichiometric needs of the algae, the algal biomass in the water body at that time would be controlled or limited by the quantity of nitrogen present in the water body. The amount of phosphorus present in the water body at that time would have no influence, in terms of limiting algal growth, since it would be present in excess quantities relative to the stoichiometric requirements of the algae. The opposite would be true if the N:P atomic ratio were greater than 16. Thus, an examination of the relative quantities of nitrogen and phosphorus in a water body at a given time, especially during the growing season, will indicate which of the two nutrients is "left over" after the other has been depleted by the algae. Clearly, the nutrient which is present in large quantities (i.e., left over) during periods of excessive algal growths is not limiting growth of the algae. Rather, the depleted nutrient is the one which would be controlling or limiting the algal growth. Other algal metabolic processes may also be occurring at the same time, such as luxury consumption of phosphorus in nitrogen-limited waters (Fitzgerald, 1969; Lee, 1973), but in general growth will be controlled by the nutrient in the water body which has been depleted, relative to the stoichiometric requirements of the algae.

Attention must be given to the forms of the nutrients available for algal and macrophyte growth, rather than to the total nitrogen or phosphorus content of the water body. Cowen and Lee (1976a) demonstrated that up to 30 percent of the particulate phosphorus in urban runoff can be converted to algal-available phosphorus (i.e., soluble orthophosphate) in about 20 days. In addition, Cowen *et al.*, (1976a) showed that up to 70 percent of the organic nitrogen in urban runoff can be converted to inorganic forms (i.e., $NH_4^+ + NO_3^- + NO_2^-$ as N) available for algal growth in 35 to 50 days. Similar findings were shown with river waters tributary to Lake Ontario (Cowen *et al.*, 1976b). However, since algal blooms are rapidly-occurring short-term events, it is the quantity of the algal-available forms of nitrogen and phosphorus present at any given time in a water body, rather than the organic fraction, or the quantities of the total phosphorus or nitrogen, that will determine which will be limiting algal growths. The available form of phosphorus in natural waters consists of the soluble orthophosphate fraction. The available nitrogen forms consist of ammonia, nitrate and nitrite.

The limiting nutrient concept, as illustrated in the N:P ratio, has been applied to the US OECD water bodies. A summary of the limiting nutrients in the US OECD water bodies, as indicated by their respective principal investigators, is presented in Table 9. In addition, the US OECD water bodies were examined for their content of available nitrogen and phosphorus and the mass ratios of inorganic nitrogen:soluble orthophosphate (as N:P) were determined. The mass ratios of N:P, rather than the atomic ratios, were computed because of ease of directly using the inorganic nitrogen and soluble orthophosphate concentrations reported by the US OECD investigators. Since the concentration volumes were the same, the inorganic nitrogen:soluble orthophosphate mass ratio was the quotient of the inorganic nitrogen concentration over the soluble orthophosphate phosphorus concentration. Incorporating the atomic weights of nitrogen and phosphorus, an N:P atomic ratio of 16:1 corresponds to an N:P mass ratio of 7.2:1. Using Selenastrum algal assays, Chiaudani and Viglis (1974) have shown that at N:P mass ratios below 5:1, nitrogen was limiting, while at N:P ratios of 10:1 or greater phosphorus was limiting. Between N:P mass ratios of 5-10 either could be limiting algal growth. In this discussion, the critical N:P mass ratio was taken as 7-8:1. A similar N:P ratio was also used by Schindler (1977) to define the limiting nutrient in his whole-lake studies in the Canadian Experimental Lakes Area. The N:P mass ratios of the US OECD water bodies are presented in Table 10.

TABLE 9. SUMMARY OF LIMITING AQUATIC PLANT NUTRIENTS IN US OECD WATER BODIES

Water Body	Limiting Aquatic Plant Nutrient[a]
Blackhawk (E)[b]	P
Brownie (E)	-
Calhoun (E)	-
Camelot-Sherwood Complex (E)	P
Canadarago (E)	P
Cayuga (M)	P (summer)[c]
Cedar (E)	-
Cox Hollow (E)	P
Dogfish (O)	-
Dutch Hollow (E)	P
George (O-M)	-
Harriet (E)	-
Isles (E)	-
Kerr Reservoir (E-M)	N-upper ends of both arms; shifting to P-limitation as one moves to lower ends of both arms
Lamb (O)	-
Meander (O)	-
Mendota (E)	P
Michigan (O-M)	P-open waters; most nearshore waters N-some nearshore waters with restricted circulation
Lower Lake Minnetonka (E→M)	P (summer)
Potomac Estuary (U-E)	N-in upper & middle portions (summer) P-in lower portion, and in upper and middle portions rest of year
Redstone (E)	P
Sallie (E)	- ("P appears not to be limiting above a certain level")

TABLE 9. (continued) SUMMARY OF LIMITING AQUATIC PLANT NUTRIENTS IN US OECD WATER BODIES

Water Body	Limiting Aquatic Plant Nutrient[a]
Sammamish (M)	P
Shagawa (E)	P
Stewart (E)	P
Tahoe (U-O)	N
Twin Lakes (E)	P (summer)
Twin Valley (E)	P
Virginia (E)	P
Waldo (U-O)	P or other?
Washington (E)	N-(in mid-1960's)
(M)	P-(prior to 1960's and in recent years)
Weir (M)	P
Wingra (E)	P

EXPLANATION:

[a]Based on investigators' estimates:
P=phosphorus-limited
N=nitrogen-limited

[b]Investigator-indicated trophic state:
E=eutrophic
M=mesotrophic
O=oligotrophic
U=ultra

[c]Period during which nutrient was specified by investigator to be limiting aquatic plant growth in water body.

Dash (-) = data not available.

Table 10. MASS RATIOS OF INORGANIC NITROGEN TO DISSOLVED PHOSPHORUS IN US OECD WATER BODIES

Water Body	Mass Ratios (Inorganic Nitrogen:Dissolved Phosphorus) Growing Season	Annual	Other
Blackhawk (E)[a] ($NH_4^+ + NO_3^- + NO_2^-$ as N)[b]	36[c]	--	26[e]
Brownie (E) ($NH_4^+ + NO_3^-$ as N)	< 5.5[d]	--	--
Calhoun (E) ($NH_4^+ + NO_3^-$ as N)	<11[d]	--	--
Camelot-Sherwood Complex (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	74[c]	--	134[e]
Canadarago (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)			
1968	10.5	19	19[f]
1969	23	22	15[f]
Cayuga (M) ($NH_4^+ + NO_3^- + NO_2^-$ as N)			
1972	117	123	--
1973	360	126	--
Cedar (E) ($NH_4^+ + NO_3^-$ as N)	11[d]	--	--
Cox Hollow (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	18[c]	--	21[e]
Dogfish (O) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	--	--	--
Dutch Hollow (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	22[c]	--	30[e]
George (O-M) ($NH_4^+ + NO_3^-$ as N)	--	25	--

Table 10 (continued). MASS RATIOS OF INORGANIC NITROGEN TO DISSOLVED PHOSPHORUS IN US OECD WATER BODIES

Water Body	Mass Ratios (Inorganic Nitrogen:Dissolved Phosphorus) Growing Season	Annual	Other
Harriet (E) ($NH_4^+ + NO_3^-$ as N)	<11[d]	--	--
Isles (E) ($NH_4^+ + NO_3^-$ as N)	5.5[d]	--	--
Kerr Reservoir (E-M) ($NH_4^+ + NO_3^- + NO_2^-$ as N)			
Roanoke Arm	22	28	50[f]
Nutbush Arm	14	11	20[f]
Lamb (O) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	--	--	--
Meander (O) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	--	--	--
Mendota (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	--	5	--
Michigan ($NH_4^+ + NO_3^- + NO_2^-$ as N)			
Near shore (M)	--	>100	--
Open waters (O)	--	170	--
Minnetonka (E→M) Nitrogen Concentrations Not Determined			
Potomac Estuary (U-E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)			
Upper Reach	2-16 (June-	--	--
Middle Reach	1- 4 Sept)	--	--
Lower Reach	1-15	--	--
Redstone (E) ($NH_4^+ + NO_3^- + NO_2^-$ as N)	38[c]	--	100[e]

Table 10 (continued). MASS RATIOS OF INORGANIC NITROGEN TO DISSOLVED PHOSPHORUS IN US OECD WATER BODIES

Water Body	Mass Ratios (Inorganic Nitrogen:Dissolved Phosphorus) Growing Season	Annual	Other
Sallie (E)			
(NH_4^++NO_3^-+NO_2^- as N)			
1972	4	3	3[f]
1973	1	--	--
Sammamish (M)			
(NO_3^-+NO_2^- as N)	60	30	--
Shagawa (E)			
(NH_4^++NO_3^-+NO_2^- as N)	8	8	8[f]
Stewart			
(NH_4^++NO_3^-+NO_2^- as N)	108[c]	--	205[e]
Tahoe (U-O)			
(NH_4^++NO_3^-+NO_2^- as N)			
1973	> 2	> 4	--
1974	> 1	--	--
East Twin			
(NH_4^++NO_3^-+NO_2^- as N)			
1971 (E)	--	27	--
1972 (E)	--	19	--
1973 (E)	--	21	--
West Twin			
(NH_4^++NO_3^-+NO_2^- as N)			
1971 (E)	--	28	--
1972 (E)	--	13	--
1973 (E)	--	14	--
Twin Valley (E)			
(NH_4^++NO_3^-+NO_2^- as N)	23[c]	--	27[e]
Virginia (E)			
(NH_4^++NO_3^-+NO_2^- as N)	7[c]	--	55[e]

Table 10(continued). MASS RATIOS OF INORGANIC NITROGEN TO DISSOLVED PHOSPHORUS IN US OECD WATER BODIES

Water Body	Mass Ratios (Inorganic Nitrogen:Dissolved Phosphorus)		
	Growing Season	Annual	Other
Waldo (U-O)			
(NH_4^++NO_3^-+NO_2^- as N)	--	< 2	--
Washington			
(NH_4^++NO_3^-+NO_2^- as N)			
1933 (E)	37	2	--
1957 (E)	21	60	--
1964 (E)	11	8	--
1971 (M)	13	30	--
Weir (M)			
(NH_4^++NO_3^-+NO_2^- as N)	2	3	--
Wingra (E)			
(NH_4^++NO_3^- as N)	17	16	--

EXPLANATION

[a]Investigator-indicated trophic state:

E = eutrophic

M = mesotrophic

O = oligotrophic

U = ultra

[b](NH_4^++NO_3^-+NO_2^-as N)=nitrogen fractions considered in N:P mass ratio calculations.

[c]Summer epilimnetic concentration.

[d]Summer surface concentration.

[e]Mean winter concentration.

[f]Spring overturn concentration.

Dash (-) indicates no data available.

Aquatic Plant Limitation in US OECD Water Bodies

Using algal assay procedures in most cases, the majority of the US OECD investigators characterized their respective water bodies as being phosphorus-limited (Table 9). The exceptions to this were ultra-oligotrophic Lake Tahoe (nitrogen-limited) and the ultra-eutrophic Potomac Estuary (nitrogen-limited in the upper and middle portions of the estuary, at least in the summer months). In addition, Lake Washington was considered nitrogen-limited in the mid-1960's, prior to diversion of domestic wastewaters; it now appears to be phosphorus-limited. Ultra-oligotrophic Lake Waldo has been shown to be phosphorus-limited in *in situ* primary productivity experiments (Powers *et al.*, 1972). However, Miller *et al.* (1974) were unable to increase algal productivity in laboratory algal assays with either phosphorus additions alone or phosphorus plus nitrogen additions. Lake Michigan is believed to be nitrogen-limited in some nearshore areas with restricted circulation, such as southern Green Bay (Lee, 1974a). The Kerr Reservoir is reported as being nitrogen-limited in its two upper arms, but shifting to phosphorus limitation as one moves toward the lower ends of both arms. Data for computing the N:P ratios were unavailable for some water bodies (e.g., Brownie, Calhoun, Cedar, Dogfish, George, Harriet, Isles, Lamb, Meander and Sallie). However, with the exception of Lakes George and Sallie, the nitrogen budgets of the above-listed water bodies were not determined by their respective US OECD investigators, implying these water bodies are phosphorus-limited. This implication may or may not be true and may reflect the biases of the investigators for these water bodies.

The inorganic nitrogen:soluble orthophosphate mass ratios of the US OECD water bodies, on both an annual and growing season basis, were presented in Table 10. Examination of this table shows that, in general, the limiting nutrient designated by the US OECD investigators for their respective water bodies was substantiated by the inorganic nitrogen:soluble orthophosphate mass ratio in the water bodies.

There were, however, a few exceptions to this observation. For example, Lakes Shagawa and Weir have both annual and growing season inorganic nitrogen:soluble orthophosphate mass ratios of 8 or less. Yet both these water bodies are phosphorus-limited, according to their respective investigators (Table 9). These discrepancies can be explained to some degree by noting when the ratios were determined. The period during which the ratio is measured clearly will influence the results obtained. This is best exemplified with the mass ratios for Lake Mendota. Its annual inorganic nitrogen: soluble orthophosphate mass ratio of 5 indicates that the lake should be nitrogen-limited. Yet, algal assay studies during the summer months clearly show Lake Mendota to be phosphorus-limited during that period. Inorganic nitrogen:soluble orthophosphate mass ratios determined during the summer months would also have indicated a phosphorus-limited water body.

Ultra-eutrophic Lake Sallie has an inorganic nitrogen:soluble orthophosphate mass ratio of 3 or less during all times of the year, indicating nitrogen limitation. According to Neel (1975), phosphorus did not seem to limit algal growth "beyond a certain point," in Lake Sallie, implying nitrogen limitation. Vollenweider (1975a; 1976a) has also reported that, even though phosphorus may initially be limiting algal growth, nitrogen may become limiting beyond a certain advanced level of eutrophication. Miller et al. (1974), studying primary productivity in 49 water bodies, reported that, in general, phosphorus limitation decreased in the water bodies as the primary productivity index increased. Vollenweider (1975a) has presented evidence that this shift to nitrogen-limitation may be due to increasing denitrification in highly eutrophic water bodies. According to Vollenweider (1975a; 1976a), this point is reached when the nitrogen residence time:phosphorus residence time ratio in the water body drops below a value of one. The nitrogen residence time:phosphorus residence time ratio , therefore, also offers a simple method for dedermining the aquatic plant growth limiting nutrient in a water body. With specific reference to Lake Sallie, another factor which should be considered in determination of its limiting nutrient is that its excessive aquatic plant growths are manifested mainly in macrophyte growths. The application of the N:P ratio concept to Lake Sallie is likely not valid because it would not account for that portion of the nutrients obtained through macrophyte root systems in the sediments. The mean inorganic nitrogen:soluble orthophosphate mass ratios in Table 10 indicate the Kerr Reservoir to be phosphorus-limited during all times of the year. However, Weiss and Moore (1975) reported that the Kerr Reservoir is initially nitrogen-limited in the upper ends of both arms, and shifts to phosphorus-limitation as one moves toward the lower ends of the two arms (Table 9). This inconsistency may be due to the fact that the upper ends of both arms of the Kerr Reservoir receive heavy sediment loads. Weiss (1977) has indicated that there may be a considerable degree of adsorption of phosphate on the clays of the heavy sediment load, producing low phosphate concentrations in the upper ends of the two arms and resulting in nitrogen-limitation. According to Weiss, this may illustrate a problem of assessing limiting nutrients in waters which have frequent incursions of Fe- and Al-rich sediments.

In summary, the use of the N:P ratio approach to estimate potential algal growth limitation by nitrogen or phosphorus requires examination of this ratio over the annual cycle. Particular attention should be given to those periods of the year when excessive planktonic algal growth causes significant water deterioration. For many water bodies this usually corresponds to the summer months, when the water body is being extensively used for recreational purposes. It is not the limiting nutrient over the annual cycle that is of importance in determining what nutrient should be considered in remedial treatment of the nutrient loading to a water body. Rather, the growing season is the period of primary concern, since algal growths during the non-growing season are seldom of consequence in terms of eutrophication control in natural waters. Also, algal growths may be limited by one nutrient during the summer months, or the growing season, and another nutrient over the annual cycle. As mentioned earlier, Lake Mendota exhibited such a trend.

Attention should be given to the forms of the nutrients available for algal growth rather than the total element content, since the algal growth in a water body at any given time is limited by the algal-available nitrogen and phosphorus forms in the water body rather than the total nutrient content. Caution should be used in estimating nitrogen or phosphorus limitation in situations where the inorganic nitrogen:soluble orthophosphate ratio in the water body is near the normal stoichiometric ratio of algae (atomic N:P ratio of 16:1 or mass ratio of 7.2:1) because both nitrogen and phosphorus concentrations are in a constant state of change. A particular ratio that exists at one time may be markedly altered by the different rates of supply of the available forms of these elements from both internal and external sources and their utilization or transformation to available forms.

Even with the above-mentioned limitations, the use of the inorganic nitrogen:soluble orthophosphate ratio represents a reasonably accurate method for determining the limiting nutrient in a water body. This chemical approach for determining the limiting nutrient in natural waters is likely to be less expensive than bioassay procedures and will yield equally meaningful results in predicting algal growth potential when interpreted properly. Further, bioassay procedures do not take into account many of the factors that would influence the availability of nitrogen and phosphorus in a water body. In addition to the results of the US OECD water bodies in promoting this approach, Lee (1973) has reported that the use of the inorganic nitrogen: soluble orthophosphate ratio in determining the limiting nutrient has also worked reasonably well in Lake Superior and the lower Madison, Wisconsin, lakes. When proper precautions are exercised in determination of this ratio, it represents a relatively simple method for making reasonable predictions as to what nutrient (i.e., nitrogen or phosphorus) is likely to limit algal growth in most natural waters.

APPROACHES USED IN US OECD EUTROPHICATION STUDY

Initial Vollenweider Phosphorus And Nitrogen Loading Diagrams

Although nutrient loading and nutrient concentration are related, it is recognized that the nutrient concentration actually controls the algal and, to some extent, macrophyte standing crops in a given water body, and thereby the eutrophication process. However, many factors directly and indirectly affect the relationship between nutrient loading and the resultant nutrient concentration (Vollenweider, 1968). Furthermore, from the point of view of eutrophication control, the nutrient loading to a water body is more easily managed than the nutrient concentration within a water body. It was the loading approach that was adopted for the US OECD eutrophication study.

Sawyer (1947) was among the first to use the concept of nutrient loading in his studies of the effects of agricultural and urban drainage and wastewaters on the fertility of the Madison, Wisconsin, lakes. He made the observation that the lake which received the greatest quantity of phosphorus and nitrogen on an areal basis experienced the most frequent and most severe algal blooms.

Rawson (1955) and Edmondson (1961) emphasized the importance of mean depth (a measure of the volume related to unit surface area) to the productivity of water bodies. In any evaluation of areal loading, this parameter took into account the degree of dilution and its effect on the nutrient concentrations in deeper bodies of water. Inclusion of mean depth in the evaluation of productivity also allowed for the role of the thermocline in influencing nutrient recycling from sediments (Stauffer and Lee, 1973).

Vollenweider (1968) quantitatively defined the relationship between nutrient loading and planktonic algal trophic response and devised a loading relationship based on these components. When Vollenweider plotted the surface area total phosphorus loading (g $P/m^2/yr$) or total nitrogen loading (g $N/m^2/yr$) versus the mean depth (m) on a log-log scale, he found that water bodies of similar trophic states appeared in the same general areas of the diagram (Figure 5). This same relationship was also derived for nitrogen loadings (Figure 6), assuming algal nitrogen requirements were related to phosphorus requirements in the ratio of 15:1 by weight. According to Vollenweider (1977), while this is about twice the mass ratio generally accepted, he felt this high N:P ratio applied to loading (not concentration) appeared to be more appropriate, and probably included effects of denitrification which reduces the available nitrogen (in terms of concentration) relative to phosphorus. Boundary loading conditions, theoretically based on Sawyer's spring overturn critical nutrient concentrations (Vollenweider, 1968; Vollenweider and Dillon, 1974), were incorporated into the diagrams, which grouped the lakes into the three standard trophic states (i.e., oligotrophic, mesotrophic and eutrophic). The lower bounary line ("permissible") designated the maximum phosphorus or nitrogen loading levels, as a function of mean depth, that a given water body could tolerate and still retain its oligotrophic character. The upper boundary line ("excessive") represented the phosphorus or nitrogen loading level, as a function of mean depth, above which a given water body would be characterized as eutrophic. The zone separating the oligotrophic and eutrophic categories represented the mesotrophic category. This was considered a transition zone between the oligotrophic and eutrophic categories.

The approximation for the permissible loading boundary condition was empirically determined to be

$$L_c(P) = 25\ \bar{z}^{0.6} \tag{1}$$

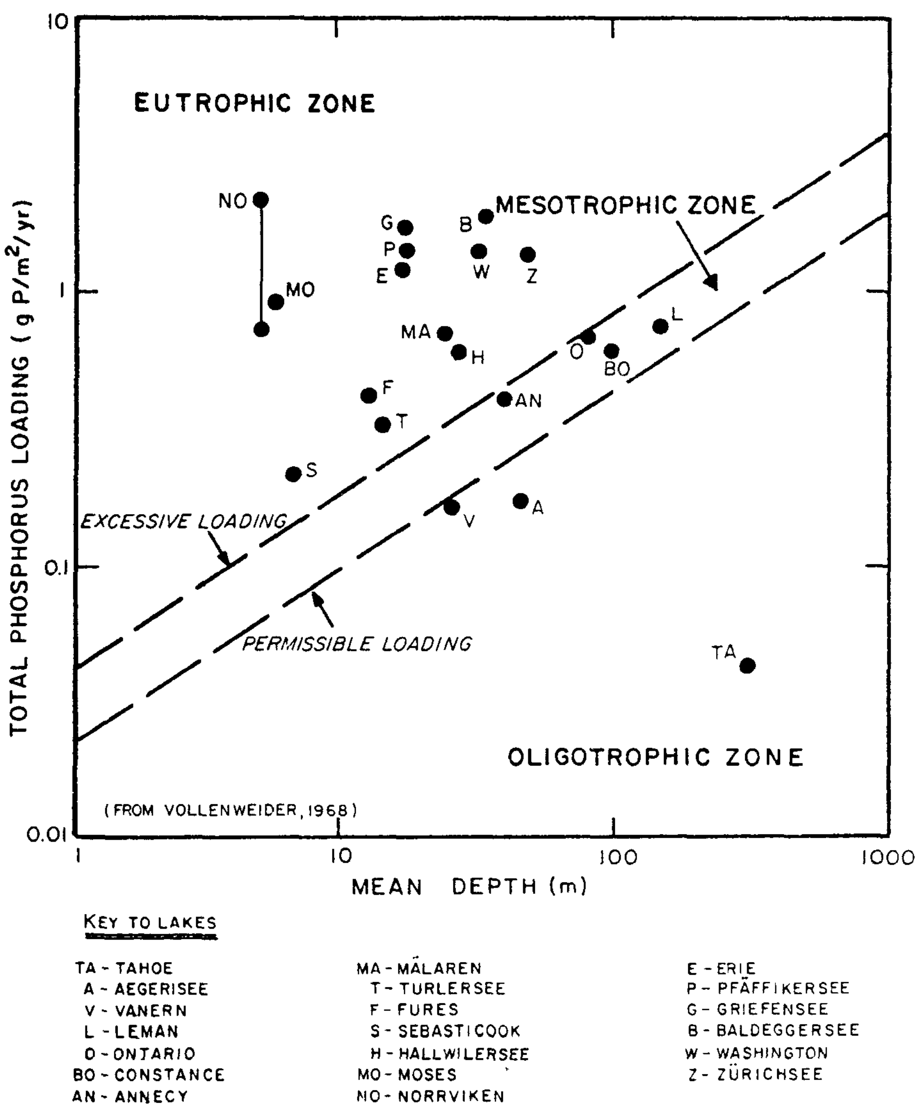


Figure 5. Vollenweider's Total Phosphorus Loading and Mean Depth Relationship.

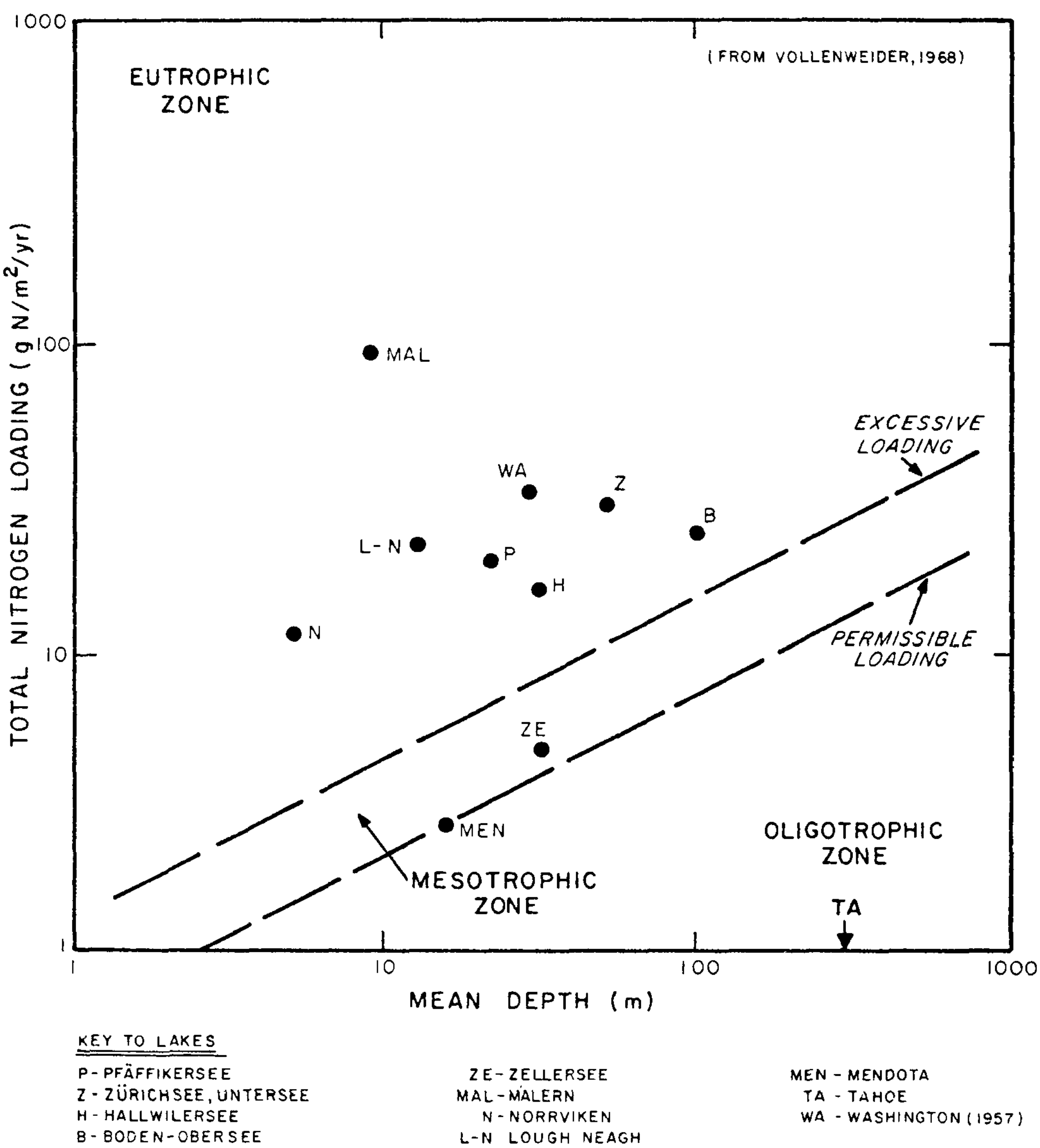


Figure 6. Vollenweider's Total Nitrogen Loading and Mean Depth Relationship.

where $L_c(P)$ = areal permissible total phosphorus loading (mg $P/m^2/yr$); and

$\bar{z}$ = mean depth (m).

The excessive loading boundary condition was considered to be approximately twice the permissible loading (Sakamoto, 1966; Vollenweider, 1968; 1976a; Dillon, 1974a; Dillon and Rigler, 1974a) as follows:

$$L(P) = 50\ \bar{z}^{0.6} \qquad (2)$$

where L(P) = areal excessive phosphorus loading (mg $P/m^2/yr$)

Assuming an N:P loading ratio of 15:1 by weight (Vollenweider, 1968), then the permissible and excessive loading lines, respectively, for nitrogen are determined by similar reasoning as:

$$L(N) = (15)\ (25 \text{ or } 50)\ \bar{z}^{0.6} \qquad (3)$$

where L(N) = areal nitrogen loading (mgN/m^2/yr).

The slope of the boundary lines indicated the greater dilution capacity of deeper water bodies, which influences their ability to assimilate more nutrients than shallower lakes without increasing their degree of fertility. A water body's relative degree of eutrophy or oligotrophy on either loading diagram was proportionate to its vertical displacement above or below the "permissible" loading line. Thus, in Figure 5, Lake Moses is relatively four times more eutrophic than Lake Sebasticook in terms of phosphorus loadings. Likewise, Lake Aegerisee is relatively more oligotrophic than Lake Vanern, based on their respective phosphorus loading and mean depth characteristics (Vollenweider and Dillon, 1974).

This model marked a significant advance in eutrophication studies and became widely accepted as a guide to the degree of eutrophy of a given water body. It was the first credible quantitative guide to "permissible" and "excessive" phosphorus and nitrogen loading levels for lakes and impoundments. That is, for most of the water bodies for which sufficient phosphorus loading data were available, the trophic state prodicted by the Vollenweider loading diagram agreed with the trophic state indicated by the standard, but arbitrary, indicators available at the time (e.g., nutrient concentrations, chlorophyll concentrations, primary productivity, Secchi depth, hypolimnetic oxygen depletion, etc.).

The Vollenweider phosphorus loading diagram was subsequently used in a number of studies to describe or predict the degree of eutrophy in various waters as a function of phosphorus loadings. For example, the International Joint Commission (1969) and Patalas (1972) used it to describe the trophic conditions of the

Great Lakes. Schindler and Nighswander (1970) used it to describe Experimental Lake 227 in their nutrient enrichment studies in northwestern Ontario. In fact, it still appears in the literature in this form even today.

However, Vollenweider (1968; 1975a) stated that his initial phosphorus and nitrogen loading diagrams were only approximate relationships and that other parameters would also have to be considered in establishing a water body's trophic status. These factors included the extent of shoreline and littoral zone, degree of nutrient mixing in the water column, internal loading from the sediments, and especially water renewal time (Vollenweider and Dillon, 1974). Vollenweider (1975a) noted that his initial model, though it worked reasonably well for hydraulic residence times of several months, did not account for the situation that two water bodies could have identical mean depths, but different hydraulic residence times. Water bodies with shorter hydraulic residence times (i.e., faster flushing rates) would also have faster cycling of water through the systems. A water body with a faster flushing rate could assimilate a larger nutrient loading, with no adverse eutrophication responses, than a slower flushing lake because of a generally faster nutrient washout which could result in a "short-circuiting" of input nutrients before they have had sufficient time to interact with the algal populations in the faster water body. Edmondson (1961; 1970a) pointed out that a lake receiving nutrients supplied in a diluted form (such as land runoff) would be affected differently than one receiving its nutrients in a concentrated form (such as domestic sewage inputs).

Dillon (1974a, 1975) was the first to report water bodies which did not fit Vollenweider's original phosphorus loading diagram scheme. In his study of the phosphorus budgets of nineteen southern Ontario lakes, he found a number of them had phosphorus loadings and mean depth characteristics which would place them in Vollenweider's eutrophic category on his loading diagram (Figure 5); yet they also had large Secchi depths, low chlorophyll concentrations and no significant hypolimnetic oxygen depletion. Dillon attributed this discrepancy to the fact that the ratios of their drainage areas to surface areas were very large. This factor and their low mean depths gave them very high flushing rates. Dillon concluded the anomalous fit of these water bodies on the Vollenweider phosphorus loading diagram was a result of their rapid flushing rates.

Vollenweider Phosphorus Loading and Nitrogen Loading Versus Mean Depth/Hydraulic Residence Time Relationships

In an attempt to allow for the effects of fast or slow flushing rates on the nutrient loading-trophic response relationships in natural waters, Vollenweider (1975a; 1976a; Vollenweider and Dillon, 1974) modified his phosphorus loading diagram to include the hydraulic residence time (i.e., water body volume/annual inflow volume). This modification was based on an input-output model involving the behavior of non-conservative substances in water bodies (Vollenweider, 1975a, Dillon, 1974b). This modification allowed the effects of hydraulic loading (as contrasted to nutrient loading) to be included along with the nutrient loading and morphometry parameters of his initial loading diagram.

Vollenweider focused his attention on modifying only the phosphorus loading diagram. He singled out phosphorus for attention because it is generally believed to be the aquatic plant nutrient most frequently controlling eutrophication in natural waters (Sawyer, 1966; Fruh et al., 1966; American Water Works Association, 1966; 1967; Vollenweider, 1968; 1975a; 1976a; Lee, 1971; 1973; Likens, 1972a; Vallentyne, 1974; Vollenweider and Dillon, 1974; US EPA, 1976a; 1976b). Furthermore, the phosphorus input to a water body is usually technologically easier to control than the nitrogen input. Much of the phosphorus supplied to water bodies is introduced by way of point sources, such as in domestic or industrial sewage. Nitrogen, while supplied from point sources, is often also introduced in large quantities from non-point (diffuse) sources, such as land runoff, precipitation, dry fallout and nitrogen fixation. These diffuse sources are usually far more difficult and expensive to control. In general, then, it is believed that the control of phosphorus loading to a water body is technically and economically more feasible than control of nitrogen loading. Consequently, Vollenweider focused on modifying his phosphorus loading diagram. Vollenweider's approach of concentrating on the phosphorus loadings to water bodies was recently given support by the general assemblies of both the International Limnological Congress and the International Ecology Congress, both of which unanimously passed resolutions recommending widespread phosphorus control as a solution to eutrophication (Schindler, 1977).

Vollenweider (1975a; 1976a) modified his relationship to include the hydraulic residence time. In this report, the hydraulic residence time is defined as the ratio of the water body volume (m^3) to the annual inflow volume (m^3/yr) and represents the lake filling time. The hydraulic residence time could also have been defined as water body volume divided by annual outflow volume since the majority of the US OECD water bodies are in the north-central and northeastern US. It is generally held that precipitation and evaporation are approximately equal over the annual cycle in these areas. Thus, the hydraulic residence times computed using the inflow volumes would presumably not be significantly different from those obtained using the outflow volumes (the importance of this parameter was recently illustrated by Piwoni _et al_. (1976) in their evaluation of the trophic state of Lake Michigan. Two different hydraulic residence times were computed, depending on whether outflow alone or outflow plus deep return flow during stratification were considered in the computations. The reader is referred to Piwoni _et al_. (1976) for a detailed discussion of this problem). Vollenweider's modification was to plot a water body's areal total phosphorus loading (g P/m^2yr) versus its ratio of mean depth (m) to hydraulic residence time (yr). This ratio was represented as $\bar{z}/\tau_\omega$. With this relationship, the critical phosphorus loading of comparable lakes is directly proportional to their mean depths, and indirectly proportional to their hydraulic residence times. The direct proportionality of the critical phosphorus loading to the mean depth relates to the dilution of the phosphorus input by the water body volume. The reciprocal proportionality of the critical phosphorus loading to the hydraulic residence time relates to the likely residence time of the input phosphorus in the water body. It was apparently Vollenweider's intent that the variables of mean depth and flushing rate be considered in this modification. However, $\bar{z}/\tau_\omega$ equals the hydraulic load, q_s (m/yr), per unit water body surface area. Thus, it appears that mean depth, as an independent parameter, is lost in part. Vollenweider's phosphorus loading versus mean depth/hydraulic residence time relationship is presented graphically in Figure 7. As with Vollenweider's original phosphorus loading diagram (Figure 5) phosphorus boundary loading lines based on Sawyer's (1947) critical nutrient concentrations, and representing the permissible and excessive phosphorus loading levels, have been drawn into Vollenweider's modified phosphorus loading diagram. According to Vollenweider (1976a), from a simple inspection of lakes plotted using this modified approach, the phosphorus loading criteria for separating oligotrophic from eutrophic lakes was as follows:

$$L_c(P) = (100)\ (\bar{z}/\tau_\omega)^{0.5} \qquad (4)$$

where $L_c(P)$ = areal permissible total phosphorus loading (mg P/m^2/yr),

$\bar{z}$ = mean depth (m), and

τ_ω = hydraulic residence time = water body volume (m^3)/annual inflow volume (m^3/yr).

As before, the excessive phosphorus loading was assumed to be equal to twice the permissible loading (Sakamoto, 1966; Vollenweider, 1975a, 1976a; Dillon, 1974a). Thus water bodies plotting above the excessive loading line are generally eutrophic while those plotting below the permissible loading line are generally oligotrophic, based on their phosphorus loadings and mean depth/hydraulic residence time characteristics. A detailed derivation of this approach is presented in Vollenweider (1975a).

It is this version of Vollenweider's model which was proposed by the US EPA (1975b, 1976a) as a basis for determining critical phosphorus loadings for US lakes and impoundments. A further modification of Vollenweider's model involves the position of the permissible and excessive loading lines in his loading diagram. This new modification, in the opinion of these reviewers,

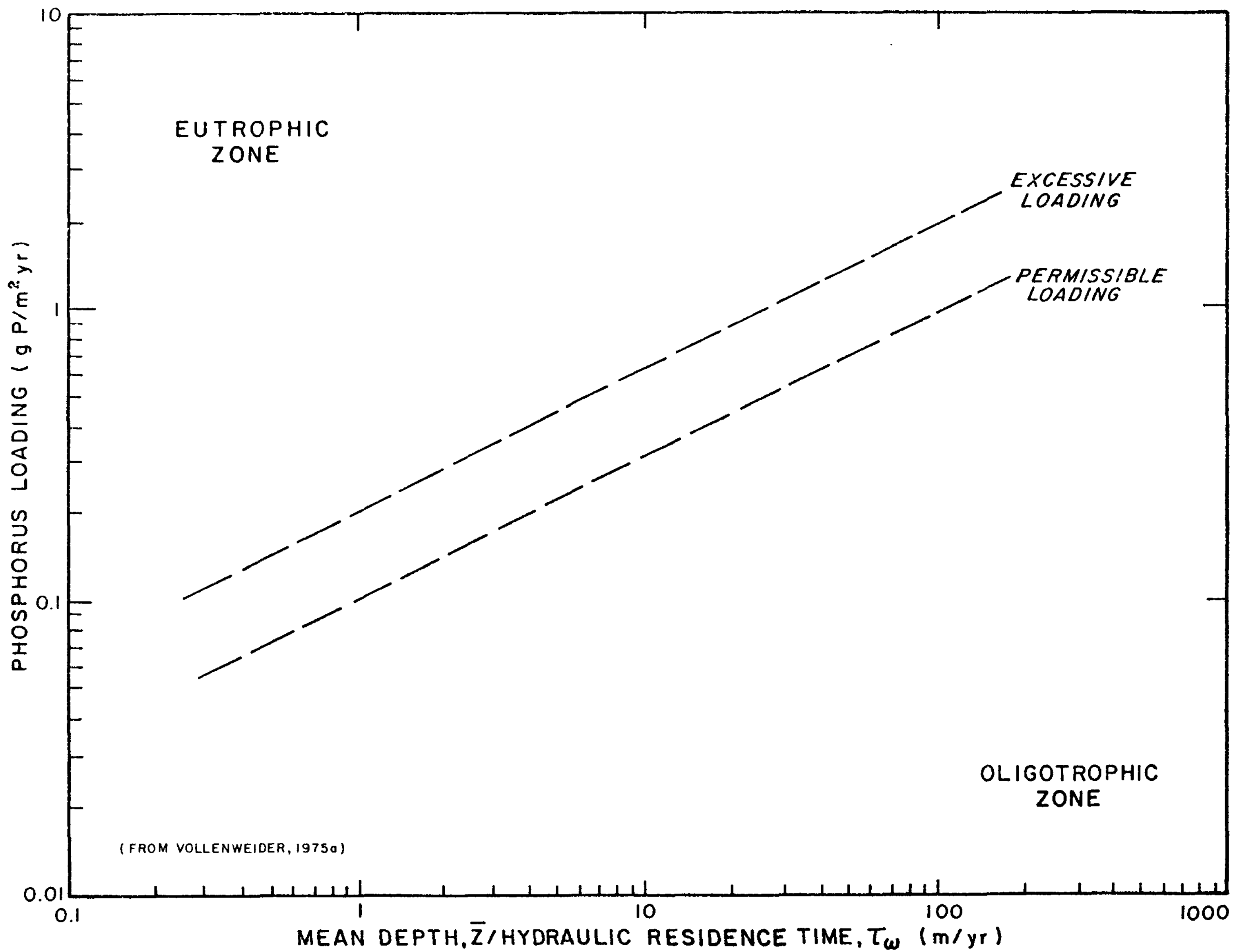


Figure 7. Initial Vollenweider Total Phosphorus Loading and Mean Depth/Hydraulic Residence Time Relationship.

marks a further refinement of Vollenweider's approach for determination of critical phosphorus loadings for lakes and impoundments. The derivation of this new modification is presented in the following section.

Based on earlier work by Biffi (1963) and Piontelli and Tonolli (1964), Vollenweider (1975a; Dillon, 1974b) developed a mass balance model for total phosphorus in natural waters. As such, it was an accountability model concerned with the balance of phosphorus between its sources and sinks. In addition to the initial mean depth parameter, this model included terms for the hydraulic residence time and a sedimentation parameter. Vollenweider's model indicated that the phosphorus dynamics of a water body can be expressed as:

$$d[P]/dt = \text{Phosphorus Load minus Outflow Loss minus Sedimentation Loss}$$

$$= (\Sigma \upsilon_j [P]_j / V) - \sigma_\rho [P] - \rho_\omega [P] \qquad (5)$$

where $[P]$ = lake total phosphorus concentration ($M\ L^{-3}$),

υ_j = flow rate of the j^{th} tributary ($L^3 T^{-1}$),

$[P]_j$ = phosphorus concentration in j^{th} tributary ($M\ L^{-3}$),

V = lake volume (L^3),

ρ_ω = hydraulic flushing rate (= annual inflow volume/lake volume) (T^{-1}), and

σ_p = phosphorus sedimentation coefficient (T^{-1}).

Vollenweider assumed a completely mixed reactor model of constant volume in which the outflow phosphorus concentration was equal to the in-lake phosphorus concentration. He further assumed the water body had equivalent inflow and outflow rates and that there was no internal loading of phosphorus to the water column from the sediments. He also assumed that phosphorus sedimentation was proportional to the phosphorus concentration in the water body, rather than to the phosphorus loading.

The time-dependent solution to this model is:

$$[P]_t = [P]_{t_o} e^{-(\rho_\omega+\sigma_p)(t-t_o)} + (\ell(P)/(\rho_\omega+\sigma_p))(1-e^{-(\rho_\omega+\sigma_p)(t-t_o)}) \quad (6)$$

The steady state solution (i.e., $t \to \infty$) to this model (Vollenweider, 1975a; 1976a) is

$$[P]_\infty = \ell(P)/(\rho_\omega + \sigma_p) \quad (7)$$

where $[P]_\infty$ = steady state total phosphorus concentration ($M\ L^{-3}$), and

$\ell(P)$ = volumnar phosphorus loading ($M\ L^{-3}\ T^{-1}$) = $\Sigma \upsilon_j [P]_j / V$

Now, $\ell(P) = L(P)/\bar{z}$, where $L(P)$ = areal total phosphorus loading and $\bar{z}$ = mean depth. Therefore, Equation 7 above becomes

$$[P]_\infty = L(P)/(\bar{z}(\rho_\omega + \sigma_p)) \quad (8)$$

Equation 8 can then be arranged as

$$L(P) = [P]_\infty \cdot \bar{z}(\rho_\omega + \sigma_p). \quad (9)$$

$[P]_\infty$ can be taken for simplicity as Sawyer's (1947) critical spring overturn phosphorus concentration of 10 mg/m^3. The hydraulic flushing rate, ρ_ω, is equal to 1/hydraulic residence time ($= 1/\tau_\omega$). The phosphorus sedimentation rate coefficient, σ_p, cannot easily be measured directly. However, Vollenweider (1975a; 1976a) has indicated as a general rule that σ_p can be approximated by

$$\sigma_p = 10/\bar{z}. \quad (10)$$

Thus, Equation 9 becomes

$$L_c(P) = [P]_c^{sp} \ \bar{z} \ (\rho_\omega + \sigma_p)$$

$$= (10 \text{ mg/m}^3)(\bar{z}/\tau_\omega + \bar{z} \ (10/\bar{z}))$$

$$= 100 + (10 \ (\bar{z}/\tau_\omega)) \qquad (11)$$

where $L_c(P)$ = areal permissible total phosphorus loading (mg $P/m^2/yr$),

$\bar{z}$ = mean depth (m),

τ_ω = hydraulic residence time (yr) = lake volume (m^3)/annual inflow volume (m^3/yr), 2nd

$[P]_c^{sp}$ = critical concentration of total phosphorus at spring overturn = 10 mg/m^3.

As with the earlier model, the excessive phosphorus loading boundary condition was considered to be approximately twice the permissible loading (Sakamoto, 1966; Vollenweider, 1968; 1976a; Dillon, 1974a: Dillon and Rigler; 1974a). Thus, the equation for the excessive loading line becomes

$$L(P) = 200 + (20 \ (\bar{z}/\tau_\omega)) \qquad (12)$$

where $L(P)$ = excessive phosphorus loading (mg $P/m^2/yr$).

These equations, theoretically based on Sawyer's (1947) critical spring overturn phosphorus concentration, serve as the basis for the modified phosphorus loading and mean depth/hydraulic residence time diagram presented in Figure 8. Vollenweider's modified phosphorus loading diagram (Figure 8) indicates that below a certain combination of mean depth and flushing, the phosphorus loading tolerance of a given water body becomes constant in spite of the fact that, based on mean depth alone, water bodies may appear to have a higher assimilation capacity. This is not indicated in his previously reported loading diagram (Figure 7). In this new modified phosphorus loading diagram, the boundary lines flatten out at $\bar{z}/\tau_\omega$ values of <2. In addition, at $\bar{z}/\tau_\omega$ values >80, the tolerable loading capacity becomes proportional to $\bar{z}/\tau_\omega$, which is contrary to what was found with his original model (Figure 7).

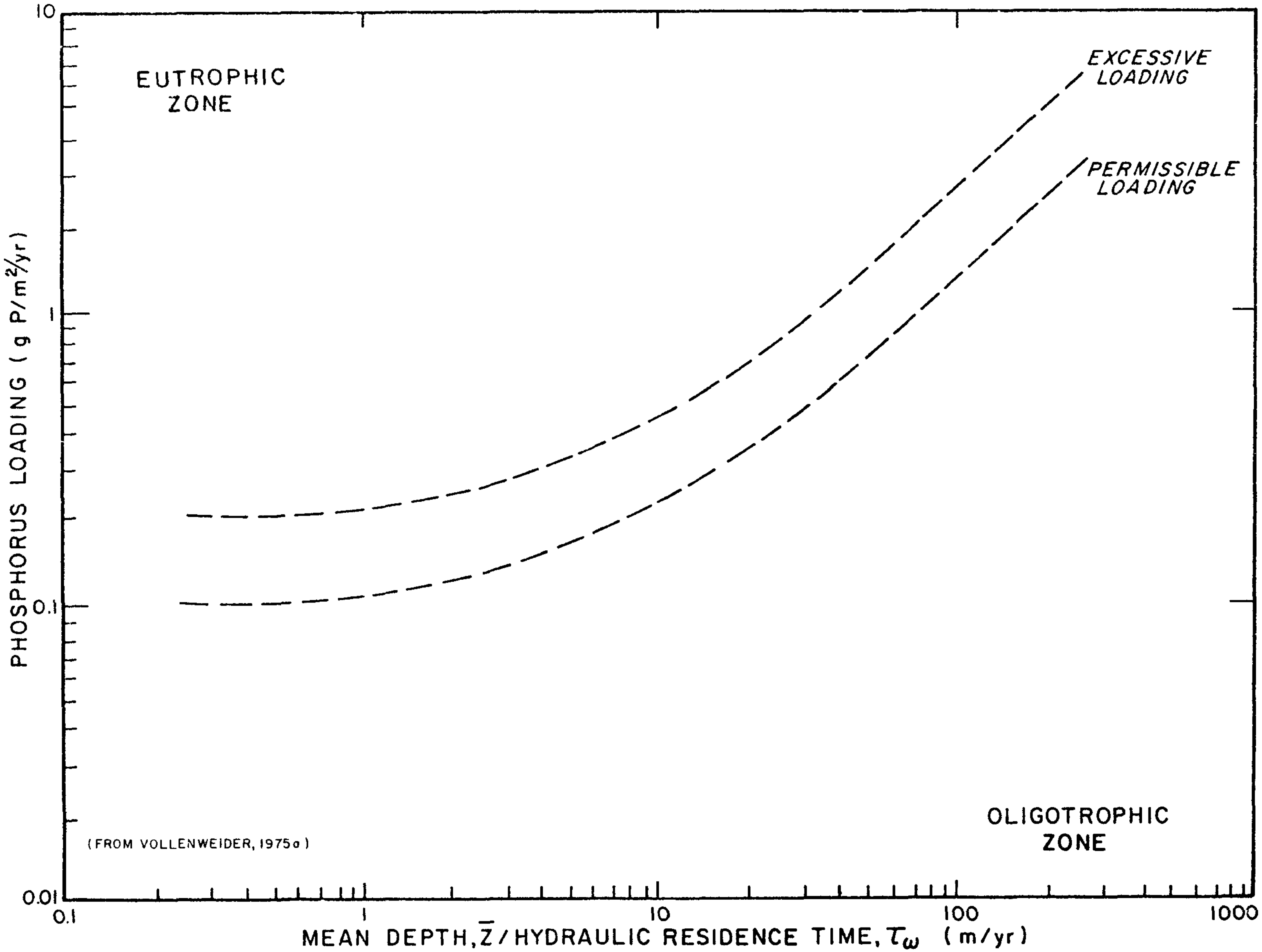


Figure 8. Modified Vollenweider Total Phosphorus Loading and Mean Depth/Hydraulic Residence Time Relationship.

A total nitrogen loading (i.e., $NH_4^+ + NO_3^- + NO_2^-$ + organic nitrogen) and mean depth/hydraulic residence time diagram has also been prepared for analysis of the US OECD eutrophication study data. The nitrogen loading diagram is identical in form to the phosphorus loading diagram except that it contains no permissible or excessive loading lines. The criteria for the positioning of the permissible and excessive boundary lines are currently being derived for water bodies which are nitrogen-limited, or which can be made nitrogen-limited with respect to aquatic plant nutrient requirements. The development of the permissible and excessive loading boundary conditions is necessary so that the type of relationship developed by Vollenweider for examining the trophic conditions of water bodies based on their phosphorus loadings and mean depth/hydraulic residence time characteristics can be applied to water bodies which are nitrogen-limited.

Emphasis on Phosphorus Loading Relationships

Vollenweider has continued to modify and improve his phosphorus loading relationships during the past several years. Moreover, others (Dillon, 1975; Larsen and Mercier, 1976) have proposed additional parameters to be considered in any evaluation of a water body's productivity and general trophic condition. These new models, to be used later in this report, are discussed in the following sections.

In all subsequent loading diagrams in this section, attention is given mainly to phosphorus loading relationships. Relationships between nutrient loadings and water body trophic response and water quality parameters are explored in later sections of this report. However, all the loading diagrams in this section relate phosphorus loadings to either influent phosphorus concentrations, chlorophyll concentrations or retention coefficients. The originators of the various loading diagrams themselves derived their loading-response relationships only for phosphorus loadings. Vollenweider (1975a) reported his concentration on phosphorus loadings stemmed from "...the relatively scant knowledge we have about other factors, e.g., nitrogen." In addition, the majority of the US OECD water bodies were characterized as being phosphorus-limited with respect to aquatic plant requirements. Consequently, all the subsequent loading diagrams refer to phosphorus loadings. It is assumed that the same relationships could be derived for nitrogen loadings. However, the originators of the subsequent loading diagrams made no attempt to do so.

Vollenweider Critical Phosphorus Loading Equations

Concurrent with his phosphorus loading diagrams, Vollenweider derived additional methods for calculating critical phosphorus loadings to water bodies. The first approximations (Vollenweider, 1976a) of the critical phosphorus loading range were given earlier in Equations 1, 4 and 11. Water bodies receiving a phosphorus loading below this permissible phosphorus loading estimate (Figures 5, 7 and 8) would be considered oligotrophic, while water bodies receiving at least twice this permissible loading would be considered eutrophic (Vollenweider, 1976a; Vollenweider and Dillon, 1974).

Vollenweider (1976a) has derived a more general relationship from Equation 9. Vollenweider (1976a; Sonzogni et al., 1976) has incorporated the concept of phosphorus residence time as a reference parameter for determining critical phosphorus loads. Vollenweider has included this parameter in this refinement of his critical phosphorus loading equation in an attempt to compensate for the loss of mean depth as an independent criterion for assessing the effects of phosphorus loading on a water body. According to Vollenweider (1976a), the concept of phosphorus residence time can be approximated in the same manner as the hydraulic residence time, or theoretical filling time, of a water body (i.e., τ_ω = water body volume/annual inflow volume). Determination of the residence time of any substance entering a water body requires only the knowledge of the loading of that substance to the water body and the mean concentration of that substance in the water body during the same time interval. Thus, for phosphorus

$$\bar{\tau}_p = [P]_\lambda / \ell(P) \qquad (13)$$

where $\bar{\tau}_p$ = phosphorus residence time (T),

$[P]_\lambda$ = mean in-lake phosphorus concentration ($M\ L^{-3}$) and

$\ell(P)$ = volumnar phosphorus loading ($M\ L^{-3}\ T^{-1}$).

Equation 13 defines the hypothetical time necessary to bring the phosphorus concentration of a water body to its present level starting from a zero phosphorus concentration in the same manner that the hydraulic residence time, as used in this report, defines the theoretical "filling time" of a water body. This same approach was used by Sonzogni et al. (1976) in development of a phosphorus residence time recovery model. This model will be discussed in a later section of this report.

However, Vollenweider (1976a) has noted that the phosphorus loading is not independent from the hydraulic loading. The only exception to this observation would be instances where the phosphorus loading is a direct input(s) of high concentration, and thus only marginally accounts for the total hydraulic loading. Therefore, Vollenweider concluded that it would be more meaningful to consider the phosphorus residence time relative to that of water.

Therefore,

$$\pi_r = \bar{\tau}_p/\tau_\omega = ([P]_\lambda/\ell(P))/(V/Q) \tag{14}$$

$$= [P]_\lambda/[P]_j$$

where π_r = phosphorus residence time relative to hydraulic residence time ($T\ T^{-1}$),

τ_ω = hydraulic residence time (T),

V = lake volume (L^3),

Q = inflow volume ($L^3\ T^{-1}$),

$[P]_j$ = mean inflow phosphorus concentration ($M\ L^{-3}$) and

$[P]_\lambda$ = mean in-lake phosphorus concentration ($M\ L^{-3}$).

In analyzing the dependence of $\bar{\tau}_p$ on τ_ω for a wide range of water bodies, Vollenweider (1976a) has noted that $\bar{\tau}_p/\tau_\omega$ is neither independent nor inversely proportional to τ_ω. Rather, $\bar{\tau}_p/\tau_\omega$ tends to decrease as τ_ω increases. He has determined that the relative phosphorus residence time depends on the hydraulic residence time by a statistical relationship which results in the following equation,

$$\pi_r = \bar{\tau}_p/\tau_\omega = \rho_\omega/(\rho_\omega + \sigma_p) \tag{15}$$

where ρ_ω = hydraulic flushing rate (T^{-1}) = $1/\tau_\omega$, and

σ_p = phosphorus sedimentation coefficient (T^{-1})

However, Vollenweider (1976a) has also noted that for lakes of less than 20 m mean depth and/or rapid flushing rates this relationship between $\bar{\tau}_p/\tau_\omega$ and τ_ω cannot be linearly extrapolated below $\tau_\omega < 1$.

An approximation which takes care of this problem is

$$\bar{\tau}_p/\tau_\omega = 1/(1 + \sqrt{\bar{z}/q_s})$$

$$= 1/(1 + \sqrt{\tau_\omega}). \qquad (16)$$

Equations 15 and 16 can then be combined as follows,

$$\bar{\tau}_p/\tau_\omega = \rho_\omega/(\rho_\omega + \sigma_p) = 1/(1 + \sqrt{\tau_\omega}). \qquad (17)$$

Equation 17 can then be solved for the sedimentation rate coefficient, σ_p, as follows,

$$\sigma_p = \sqrt{\tau_\omega}/\tau_\omega = \sqrt{\bar{z}/q_s}/\tau_\omega \qquad (18)$$

If this estimate of σ_p is inserted into Equation 9, a more generalized relationship is obtained for determining critical phosphorus loads which holds over the entire spectrum of combinations of mean depth and hydraulic loadings. This relationship is derived as follows,

$$L_c(P) = [P]_\infty \, \bar{z}(\rho_\omega + \sigma_p)$$

$$= [P]_c^{sp} \, [(\bar{z}/\tau_\omega) + (\bar{z}/\tau_\omega)\sqrt{\bar{z}/q_s}]$$

$$= 10 \cdot q_s \, (1 + \sqrt{\bar{z}/q_s}) \qquad (19)$$

where $[P]_c^{sp}$ = Sawyer's (1947) critical spring overturn phosphorus concentration = 10 mg/m^3,

$\bar{z}$ = mean depth (m),

τ_ω = hydraulic residence time (yr), and

q_s = hydraulic loading (m/yr) = $\bar{z}/\tau_\omega$.

This equation expresses the phosphorus loading tolerance in terms of the morphometry of the water body (condensed into the term of mean depth, $\bar{z}$), and the hydrologic properties of the water body (expressed as hydraulic loading, q_s). Thus, in principle, the phosphorus loading tolerance of a water body can be considered as a function of its mean depth and hydraulic loading (Vollenweider, 1976a).

This relationship has been developed by Vollenweider into the form of two equivalent diagrams (Figures 9 and 10). In Figure 9, the permissible phosphorus loading, $L_c(P)$, is plotted against mean depth and parameterized as a function of the hydraulic loading, q_s. In Figure 10, $L_c(P)$ is plotted against the hydraulic load and parameterized as a function of mean depth, $\bar{z}$.

Vollenweider Phosphorus Loading Characteristics and Mean Epilimnetic Chlorophyll a Relationship

Equations 8 or 19 can be rewritten in terms of the relationship between the phosphorus loading and the resultant phosphorus concentration in the water body, rather than in terms of critical phosphorus loading levels.

Recalling that $\rho_\omega = 1/\tau_\omega$, $\sigma_p = \sqrt{\tau_\omega}/\tau_\omega$ and $\tau_\omega = \bar{z}/q_s$, Equation 8 can be rearranged as follows:

$$[P]_\infty = (L(P)/q_s)\ (1/(1 + \sqrt{\bar{z}/q_s})) \qquad (20)$$

Equation 20, therefore, relates the predicted in-lake phosphorus concentration (assuming a steady-state condition) to an equivalent expression involving the phosphorus loading as modified by the hydraulic load. According to Vollenweider (1975a; 1976a) $L(P)/q_s$ represents the average inflow phosphorus concentration. This useful relationship will be used in a later portion of this report to check the phosphorus loads reported for the US OECD eutrophication study water bodies.

Several authors (Sawyer, 1947; Sakamoto, 1966; Dillon, 1974a; Dillon and Rigler, 1974a; Bachmann and Jones, 1974; Jones and Bachmann, 1976) have shown that a relationship exists between the phosphorus concentration at spring overturn and the mean chlorophyll concentrations in a water body during the following summer growing season. Since a positive correlation has been shown to exist between spring overturn phosphorus concentration and average summer chlorophyll concentration in a water body, it is logical to assume a positive correlation may exist between phosphorus loading and average chlorophyll concentrations. Vollenweider demonstrated such a correlation between phosphorus loadings and chlorophyll concentrations at the 1975 North American Project Meeting in Minneapolis. He plotted the phosphorus loadings of a water body,

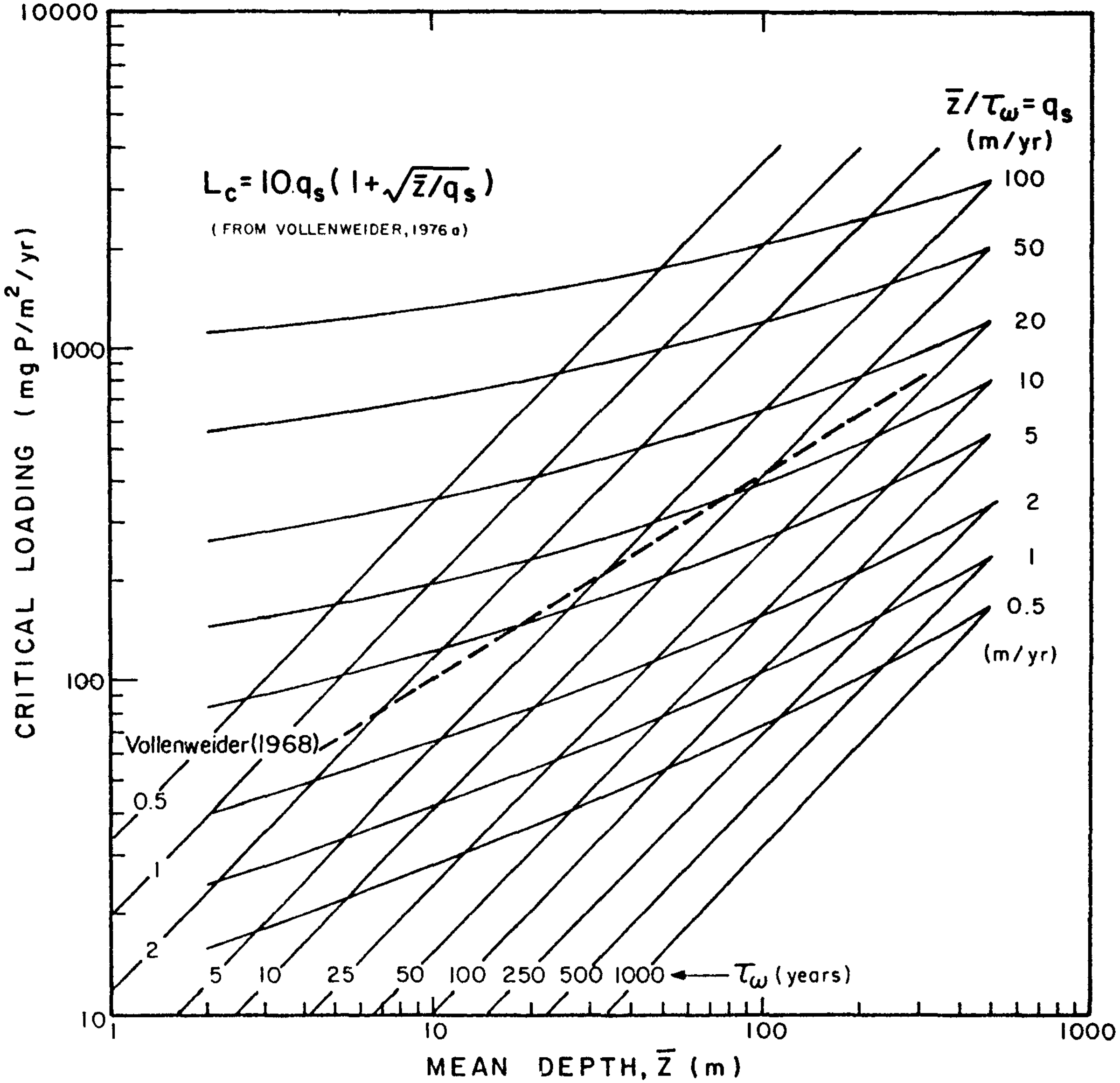


Figure 9. Vollenweider Critical Phosphorus Loading and Mean Depth Relationship.

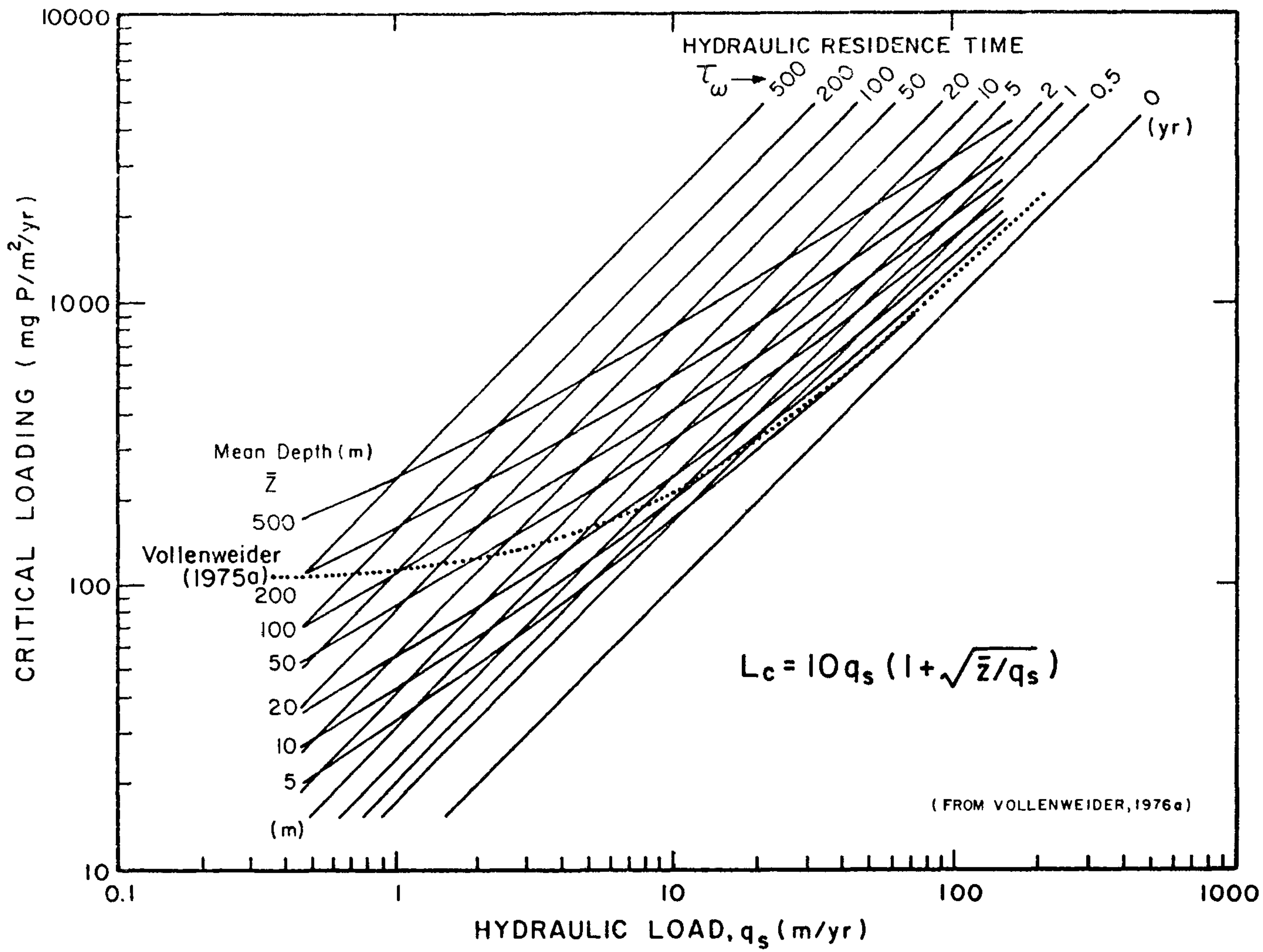


Figure 10. Vollenweider Critical Phosphorus Loading and Hydraulic Loading Relationship.

as manifested in the phosphorus loading characteristics term $(L(P)/q_s)(1/(1+\sqrt{\bar{z}/q_s}))$ in Equation 20, and the mean epilimnetic chlorophyll *a* concentration of the water body. Even though the chlorophyll *a* concentrations consist of a mixture of annual and summer average values, Vollenweider showed a definite relationship ($r = 0.87$) between the phosphorus loading characteristics of a water body and its average epilimnetic chlorophyll *a* concentration. Vollenweider's resulting loading diagram is presented in Figure 11. This diagram includes confidence intervals for prediction of chlorophyll concentrations in a water body as a function of its phosphorus loading, as modified by its hydraulic loading. The reader is reminded that since the phosphorus loading characteristic term is equivalent to the predicted mean in-lake phosphorus concentration (Equation 20), assuming a steady state condition, Vollenweider is, in effect, relating chlorophyll *a* concentrations to total phosphorus concentrations in the same manner as other researchers (Sakamoto, 1966; Dillon, 1974a; Jones and Bachmann, 1976). However, Vollenweider's contribution was to provide a phosphorus loading term, modified by hydraulic loading, which was equivalent to the predicted in-lake phosphorus concentration (Equation 20). Thus, Figure 11 indicates the relationship between predicted in-lake phosphorus concentration, as well as the phosphorus loading characteristics, and the mean epilimnetic chlorophyll *a* concentrations in a water body. In this manner, chlorophyll *a* concentrations can be related to phosphorus loadings, as well as to mean phosphorus concentrations. Larsen and Mercier (1976) used the same phosphorus loading relationship in shifting emphasis from phosphorus loadings to influent phosphorus concentrations. This will be considered in a later section of this report.

It should be noted that the response of a water body to a reduction in phosphorus loading will not be an immediate accompanying reduction in the chlorophyll concentration of the water body. Rather, there will be a "lag period" during which the phosphorus concentrations, and hence, chlorophyll *a* concentrations, in the water body are adjusting to the new phosphorus loadings. When the water body has reached a new equilibrium condition with respect to its phosphorus concentrations, then the loading diagram (Figure 11) can validly be used to predict the expected chlorophyll biomass in the water body. Vollenweider (1976a) has demonstrated this lag phenomenon with data from Lake Washington. This concept is examined by Sonzogni *et al.* (1976) in their phosphorus residence time model, and will be explored further in a later section of this report.

Dillon Phosphorus Loading-Phosphorus Retention and Mean Depth Relationship

Dillon (Vollenweider and Dillon, 1974; Dillon, 1975) was one of the first to point out one of the omissions of Vollenweider's

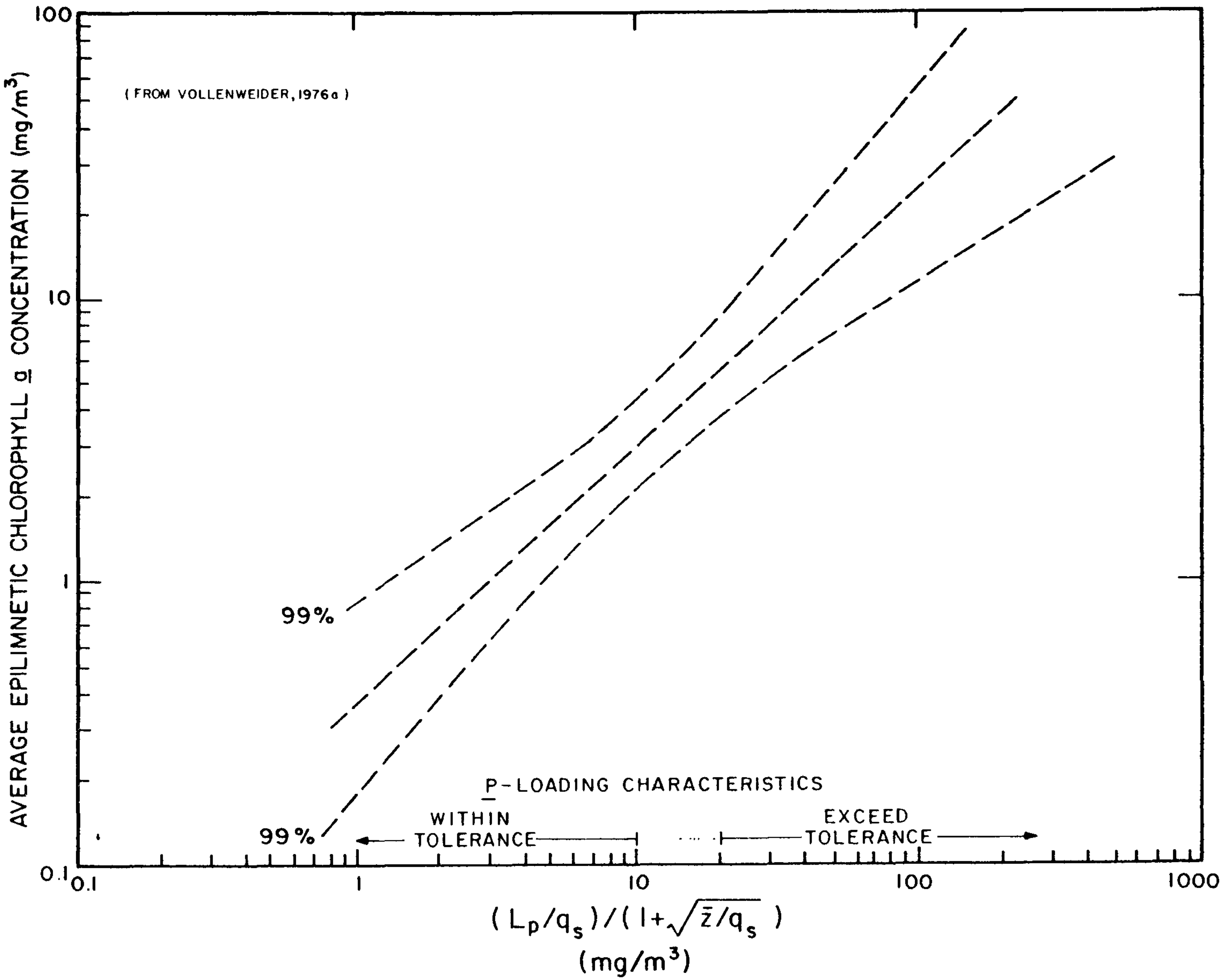


Figure 11. Vollenweider Phosphorus Loading Characteristics and Mean Chlorophyll a Relationship

original phosphorus loading diagram (Figure 5). Because flushing rate and hydraulic residence time, as well as phosphorus loading and mean depth, play a part in determining the relative degree of fertility of a water body, Dillon attempted to include these parameters in a formulation of his own.

Dillon derived his model from Vollenweider's original phosphorus mass balance model, as indicated in Equation 5. The steady state solution to Vollenweider's model (Equation 8) was shown to be $[P]_\infty = L(P)/(\bar{z}/\tau_\omega + \bar{z}/\sigma_p)$. However, as mentioned earlier, measurement of σ_p is very difficult and only indirectly obtainable. Consequently, using the same assumptions as were used to derive the model, Dillon (1975; Dillon and Rigler, 1974a) derived an alternate parameter, the phosphorus retention coefficient, R(P), which can be shown to have a functional relationship to Vollenweider's phosphorus sedimentation rate coefficient, σ_p. Dillon (1975; Dillon and Rigler, 1974a) has indicated that R(P) can be approximated, assuming a steady state condition, as

$$R(P) = 1 - (\Sigma q_o [P]_o / \Sigma q_i [P]_i) \qquad (21)$$

where q_o = outflow volume (m^3/yr),

q_i = inflow volume (m^3/yr),

$[P]_o$ = outflow concentration (mg/m^3), and

$[P]_i$ = inflow concentration (mg/m^3).

Thus R(P) represents the fraction of the phosphorus input which is retained in the sediments of the water body (i.e., the fraction of the inflowing phosphorus which sediments annually). Conversely, 1-R(P) is the fraction of inflowing phosphorus not retained in the water body (i.e., it is lost by way of outflow). Kirchner and Dillon (1975) have demonstrated that R(P) was highly correlated with the areal water loading. Using multiple regression analysis they have produced a regression equation for predicting R(P) which is very similar to the value predicted on theoretical grounds (Snodgrass, 1974; Snodgrass and O'Melia, 1975). Chapra (1975) has presented an interpretation of the high correlation found between R(P) and the areal water loading and derived an alternate method of determining R(P) as follows,

$$R(P) = \nu/(q_s + \nu) \qquad (22)$$

υ = apparent settling velocity of total phosphorus = $\propto \upsilon'$,

q_s = areal water load = Q/A,

$\propto$ = fraction of total phosphorus represented by settleable particulate phosphorus,

υ' = settling velocity of settleable particulate phosphorus,

Q = lake discharge volume, and

A = water body surface area.

Regardless of how it is determined, Dillon (1975; Dillon and Rigler, 1974a; 1974b; 1975) has shown that when R(P) is calculated and substituted into Equation 8, the equation can be rewritten as

$$[P]_\infty = (L(P)\ (1-R(P)))/\bar{z}\ \rho_\omega \qquad (23)$$

This equation attempts to consider the effects of phosphorus retention, as well as flushing rate and phosphorus loading, on the degree of fertility of a water body. It should be noted that the external loading, L(P), is in effect lost as an independent parameter since, by definition, L(P) (1-R(P)) is that part of the external phosphorus loading which is lost through the outlet. Thus, L(P) (1-R(P)) can be defined as the average outflow concentration. Therefore, in the strictest sense, Dillon's model cannot be used for defining loading tolerances as long as there is no valid model available for determining R(P). Dillon (Kirchner and Dillon, 1975) and Chapra (1975) have attempted to derive an independent and valid model for R(P), as was mentioned earlier. The effect of mean depth as an independent parameter is again partially lost since $\rho_\omega = 1/\tau_\omega = Q/V = Q/(A \cdot \bar{z})$, where A = surface area of water body. Therefore, $\bar{z}\ \rho_\omega = \bar{z}\ (Q/(A \cdot \bar{z}))$ = Q/A. As indicated earlier, Q/A is the areal water loading. Thus, Equation 23 defines the steady state phosphorus concentration of a water body as directly proportional to the product of the phosphorus loading and outflow phosphorus loss (i.e., "average outflow concentration"), and inversely proportional to the areal water loading. The areal water loading is equivalent to the hydraulic loading, q_s (i.e., $q_s = Q/A = Q/(V/\bar{z}) = \bar{z}(V/Q) = \bar{z}\ \rho_\omega = \bar{z}/\tau_\omega$).

Inclusion of the factor (1-R(P), therefore, accounts for one more source of variation in determining a water body's trophic status. Dillon (1975; Vollenweider and Dillon, 1974) prepared a loading diagram upon which is plotted $(L(P)(1-R(P)))/\rho$ versus $\bar{z}$ (Figure 12). Boundary lines representing phosphorus concentrations of 0.01 mg/l and 0.02 mg/l (Sawyer, 1947; Sakamoto, 1966; Dillon, 1975) can be drawn on the diagram. These boundary lines correspond to Vollenweider's "permissible" and "excessive" boundary conditions (Figures 7 and 8). Water bodies below the 0.01 mg/l phosphorus concentration line are considered oligotrophic and those above the 0.02 mg/l phosphorus concentration line are considered eutrophic. The transition zone between the 0.01 and 0.02 mg/l phosphorus concentration lines is considered the mesotrophic zone.

In Dillon's model, the trophic categorization of a water body is based on measurement of the water body's phosphorus concentration, rather than its phosphorus loading. This line of reasoning is consistent with the view mentioned earlier that the nutrient concentration, rather than nutrient loading, determines a water body's degree of eutrophication.

Dillon's model has its quantitative basis in the same simple nutrient budget model as does Vollenweider's model (Vollenweider, 1975a). In addition, it is a simple method for predicting phosphorus concentrations in water bodies. If these concentrations can, in turn, be related to water quality parameters that reflect a water body's trophic condition (e.g., chlorophyll concentrations, productivity, Secchi depth, etc.), then measurement of phosphorus concentration becomes a very convenient way to define or predict trophic status. As mentioned earlier, Dillon (1974a; Dillon and Rigler, 1974a) and other workers (Sakamoto, 1966; Jones and Bachmann, 1976) found such a correlation between phosphorus concentration at spring overturn and predicted average summer chlorophyll _a_ concentration.

Larsen and Mercier Influent Phosphorus And Phosphorus Retention Relationship

Larsen and Mercier (1976) shifted emphasis from phosphorus loadings to average influent phosphorus concentrations as a measure of trophic state. They described the average phosphorus concentration in a water body as a function of the relationship between the mean influent phosphorus concentration and the water body's ability to assimilate the influent phosphorus. Their model, like Dillon's model, was derived from the steady state solution of a simple phosphorus mass balance model such as

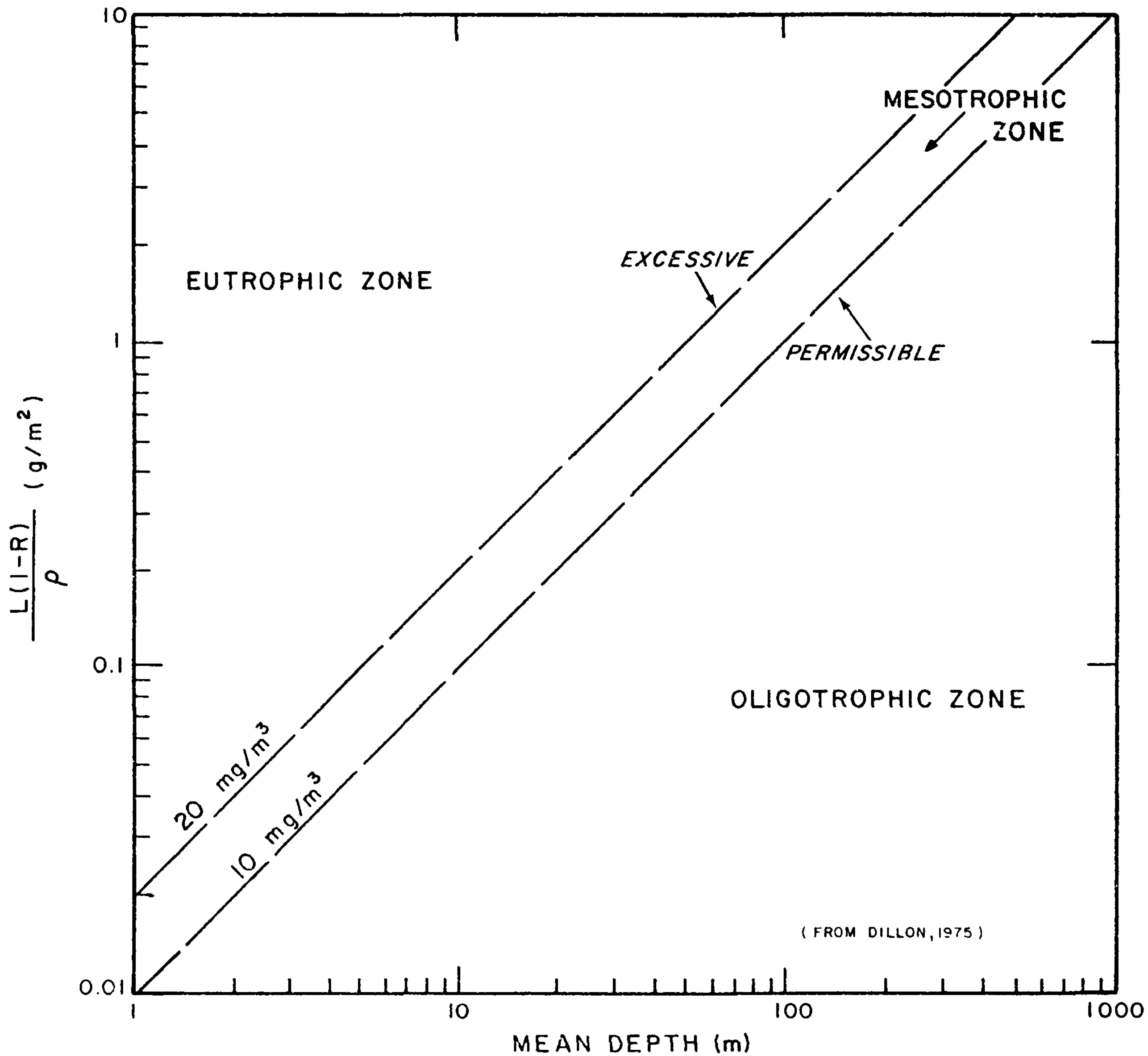


Figure 12. Dillon Phosphorus Loading-Phosphorus Retention and Mean Depth Relationship.

presented by Vollenweider (Equation 8) (1975a). Recalling that $\rho_\omega = 1/\tau_\omega$ and $\bar{z}/\tau_\omega = q_s$, Equation 23 can be rewritten as

$$[P]_\alpha = L(P)\ (1-R(P))\ /\ (\bar{z}/\tau_\omega)$$

$$= (L(P)/q_s)\ (1-R(P))$$

$$= \overline{[P]}\ (1-R(P)) \qquad (24)$$

where $\overline{[P]}$ = influent phosphorus concentration (mg/m^3) = $L(P)/q_s$, and

$1-R(P)$ = fraction of phosphorus input not retained by sediments.

This relationship is identical to that of Dillon (Equation 23) since $L(P)/\bar{z}\ \rho_\omega = L(P)/q_s = \overline{[P]}$. Thus Larsen and Mercier's relationship relates the steady state phosphorus concentration of a water body to the product of the influent phosphorus concentration and the fraction of the phosphorus input which is not sedimented.

Larsen and Mercier's (1976) relationship (Equation 24) between water body steady state in-lake phosphorus concentration and phosphorus retention is identical to that relationship implicitly indicated earlier in Vollenweider's equation for determining the critical phosphorus loading for a water body, based on its mean depth and hydraulic load (Equation 19). According to Vollenweider (1975) and Larsen and Mercier (1976), $R(P) = 1/(1 + \sqrt{\rho_\omega})$. Therefore, Equation 19 can be shown to be equivalent to Equation 24 as follows:

from Equation 19

$$L_c(P) = 10 \cdot q_s(1 + \sqrt{\bar{z}/q_s})$$

Rearranging,

$$10 = (L_c(P)/q_s)\ (1/(1 + \sqrt{z/q_s})).$$

Since $L_c(P)/q_s = \overline{[P]}$, and $\bar{z}/q_s = \tau_\omega$, then

$$10 = \overline{[P]}\ (1/(1 + \sqrt{\tau_\omega})).$$

from Equation 24

$$[P]_\infty = \overline{[P]}\ (1-R(P)).$$

Taking, for simplicity, Sawyer's (1947) spring overturn critical phosphorus concentration of 10 mg/m^3 as $[P]_\infty$, and recalling $R(P) = 1/(1 + \sqrt{\rho_\omega})$ and $\rho_\omega = 1/\tau_\omega$,

$$10 = \overline{[P]}\ (1-(1/(1 + (1/\sqrt{\tau_\omega}))))$$

$$= \overline{[P]}\ (1/(1 + \sqrt{\tau_\omega})).$$

The same results are obtained using either equation.

Larsen and Mercier (1976) prepared a phosphorus diagram to show the relationship between a water body's influent phosphorus concentration and its phosphorus retention capacity, as illustrated in Figure 13. Curves delineating trophic states can be drawn on Larsen and Mercier's diagram in a manner analogous to the method in which they have been plotted on the previous loading diagrams. Thus, this diagram can be used to determine the reduction of a water body's influent phosphorus concentration necessary to improve its trophic condition. Since Larsen and Mercier's diagram attempts to relate trophic state and in-lake phosphorus concentrations, it can also be related to other parameters of water quality (e.g., chlorophyll concentrations, productivity, Secchi depth, etc.). For the same values of L(P), ρ_ω $\bar{z}$, and R(P), the relative positions of lakes plotted on Dillon's loading diagram (Figure 12) would be identical to those on Larsen and Mercier's diagram (Figure 13) because both diagrams estimate the same property, namely in-lake steady state phosphorus concentration, from the same variables.

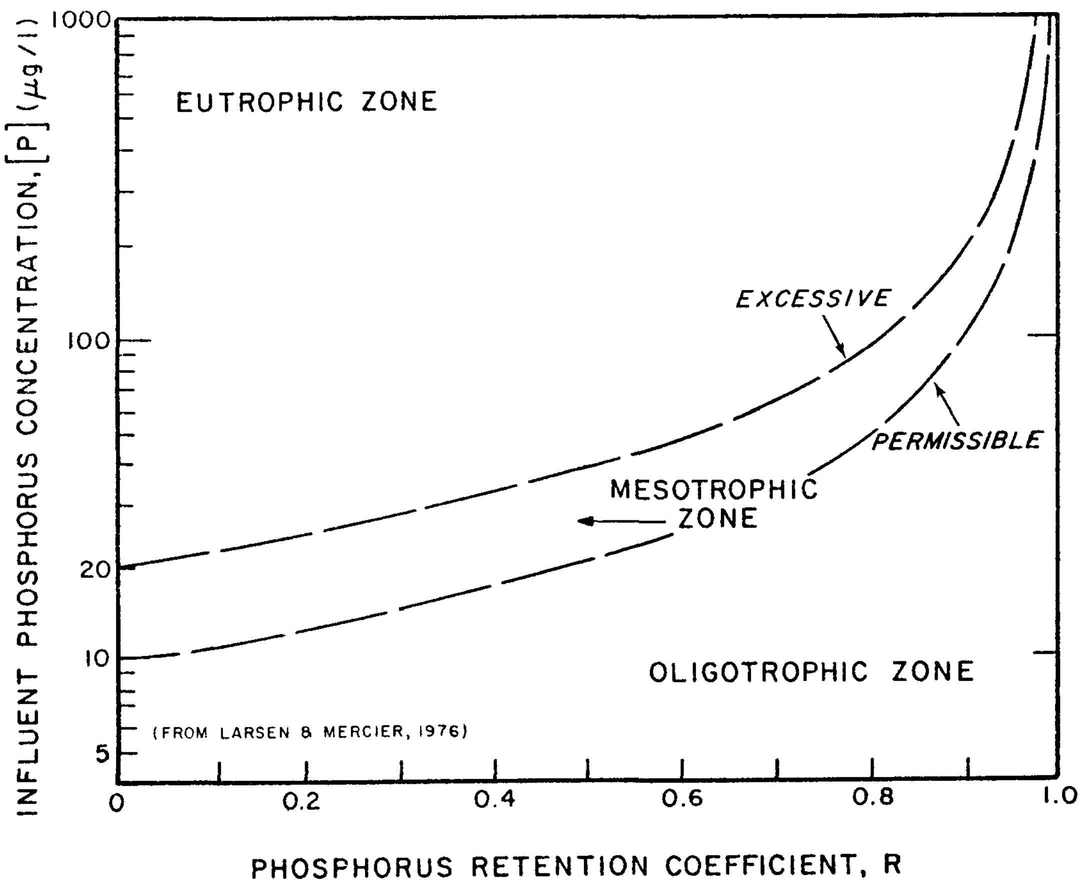


Figure 13. Larsen and Mercier Influent Phosphorus and Phosphorus Retention Relationship.

SECTION VI

RESULTS OF THE INITIAL ANALYSIS OF THE US OECD EUTROPHICATION STUDY DATA

The overall approach utilized in the US OECD eutrophication study involved giving each of the US investigators a small amount of funds to develop a report covering the topics listed in Appendix I. Each investigator prepared a preliminary draft report which was made available to all the other US OECD investigators in the spring of 1974. During the remainder of 1974 and early 1975 each investigator revised his report so that it conformed to the form outlined in Appendix I. The US EPA limited each report to approximately 20 typewritten pages. These reports were submitted to the US EPA on or about July 1, 1975. At that time they were made available to the authors of this report for examination.

This section of this report involves a detailed examination of the information provided on sampling, analytical and other methodology used by the US OECD investigators to generate the summary data sheet for their respective water bodies as presented in Appendix II. This section also examines the various methods used by the US OECD investigators to estimate nutrient load-lake or impoundment trophic response relationships. Particular attention was given to the nutrient loading estimates as they are applied in the loading diagrams developed by Vollenweider and others for establishing critical phosphorus loadings and trophic state associations for lakes and impoundments.

SAMPLING AND MEASUREMENT METHODOLOGIES

The US OECD water bodies were examined both for nutrient flux and trophic response. A water body's trophic response was measured by a variety of physical, chemical and biological parameters, as outlined in the Final Report Outline (Appendix I) and summarized in the investigators' Summary Sheets (Appendix II). The various response parameters deemed essential or desirable in the OECD eutrophication study (Table 2) had been agreed upon prior to the initiation of the study. However, most of the US OECD water bodies had been extensively studied prior to initiation of the US OECD eutrophication study. In most cases the goals of the prior studies were often different from those of the US OECD eutrophication study. Also, the sampling and analytical methodologies employed in the earlier studies were often different from those suggested and outlined by the OECD Water Management Sector Group prior to initiation of the OECD eutrophication study. A summary of the analytical methodologies used by the US OECD investigators in determining the major response parameters is presented in Table 11, while the sampling methodologies are presented on the Summary Sheets (Appendix II). Examination of Table 11 indicates that while the US EPA (US EPA, 1971; 1973d; 1974b) and Standard Methods (APHA et al., 1971) served as the major sources of analytical methodology, there was still a wide variety of methods used by the US OECD investigators to determine various parameters. In addition, the sampling regimes, including sampling depths, frequencies, and durations, varied widely among investigators. For example, the "mean" value for a given parameter was biased both by the period of sampling and the frequency with which the water body was sampled. Some water bodies were sampled at regular intervals, while others were sampled only during the ice-free period or during a specific month of the year. Also, some water bodies were sampled at many depths while others were sampled only at a few depths. Any sampling and/or analytical errors were also incorporated into determination of the mean values. The result of these variations is that direct comparison of values between water bodies is often not valid. Standardization of all sampling methodologies and analytical procedures is necessary before such direct comparison of trophic response parameters between US OECD water bodies is valid.

NUTRIENT LOAD CALCULATION METHODOLOGIES

The usefulness of the various Vollenweider phosphorus loading relationships, as well as the relationships developed by Dillon (1975) and Larsen and Mercier (1976), for establishing critical phosphorus loading rates and trophic state associations is dependent upon the accuracy of the water body's phosphorus loading estimates. Consequently, before reviewing the nutrient load-trophic response relationships found in the US OECD eutrophication study, it is appropriate to review the various methods used by the US OECD investigators to calculate the parameters necessary for the various nutrient loading diagrams derived in the previous section.

A summary of the methods used to estimate the nutrient loadings to the US OECD water bodies is presented in Table 12. Examination of this table indicates a variety of different methods were employed by the US OECD investigators to estimate the nutrient loadings. An attempt was made to clarify and standardize these various methodologies. Such standardization is necessary so that the loading estimates may be directly comparable between water bodies in the US OECD eutrophication study. However, the

Table 11a. ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - PHOSPHORUS AND NITROGEN CONCENTRATIONS[a]

Water Body	Dissolved Phosphorus	Total Phosphorus	Ammonia	Nitrate	Nitrite
Blackhawk	Ascorbic Acid Method (APHA *et al.*, 1971)	Persulfate digestion followed by Ascorbic Acid Method (APHA *et al.*, 1971)	Phenate Method (APHA *et al.*, 1971)	-	-
Brownie	-	-	-	-	-
Calhoun	-	-	-	-	-
Camelot-Sherwood Complex	Ascorbic Acid Method (APHA et al., 1971)	Persulfate digestion followed by Ascorbic Acid Method (APHA et al., 1971)	Phenate Method (APHA *et al.*, 1971)	-	-
Canadarago	Murphy and Riley Method (1962)	Concentrated Sulfuric Acid & Potassium Persulfate digestion, followed by Murphy & Riley Method (1962)	Direct Nesslerization (APHA *et al.*, 1971)	Mullin and Riley Procedure (1955) if < 30 ug/l; AutoAnalyzer if > 30 ug/l	AutoAnalyzer or APHA *et al.*(1971)
Cayuga	-	-	-	-	-
Cedar	-	-	-	-	-
Dogfish	Not determined	Potassium Persulfate digestion, followed by Phosphomolybdate/ Stannous Chloride Reduction (APHA *et al.*, 1971)	APHA *et al.* (1971)	Cadmium Reduction Method (APHA *et al.*, 1971)	APHA *et al.* (1971)

Table 11a (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - PHOSPHORUS AND NITROGEN CONCENTRATIONS[a]

Water Body	Dissolved Phosphorus	Total Phosphorus	Ammonia	Nitrate	Nitrite
Dutch Hollow	Ascorbic Acid Method (APHA et al., 1971)	Persulfate digestion, followed by Ascorbic Acid Method (APHA et al., 1971)	Phenate Method (APHA et al., 1971)	-	-
George	-	-	-	-	-
Harriet	-	-	-	-	-
Isles	-	-	-	-	-
Kerr Reservoir	Automated Phosphomolybdate/ Stannous Chloride Reduction (APHA et al., 1971). Ascorbic Acid Reduction Method used after July, 1975 (APHA et al., 1971)	Potassium Persulfate& Sulfuric Acid digestion, followed by Ascorbic Acid reduction Method (APHA et al., 1971)	Automated Phenolate Method with Technicon AutoAnalyzer I (US EPA, 1971)	Automated Hydrazine Reduction with Technicon AutoAnalyzer I (US EPA, 1971) from 1966-1975; Cu/Cd Reduction (US EPA,1974b) after July, 1975	
Lamb	Not determined	Potassium Persulfate digestion, followed by Phosphomolybdate/ Stannous Chloride Reduction (APHA et al., 1971)	APHA et al. (1971)	Cadmium Reduction Method (APHA et al. 1971)	APHA et al. (1971)

Table 11a (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - PHOSPHORUS AND NITROGEN CONCENTRATIONS[a]

Water Body	Dissolved Phosphorus	Total Phosphorus	Ammonia	Nitrate	Nitrite
Meander	Not determined	Potassium Persulfate digestion, followed by Phosphomolybdate/ Stannous Chloride Reduction (APHA *et al.*, 1971)	APHA *et al.* (1971)	Cadmium Reduction Method APHA *et al.* 1971)	APHA *et al.* (1971)
Mendota	Analytical procedures outlined in Lee (1966)				
Michigan	Analytical procedures outlined in Rousar (1973)				
Minnetonka	Phosphomolybdate/ Ascorbic Acid Reduction (APHA *et al.*, 1971)	Persulfate digestion, followed by Phosphomolybdate/Ascorbic Acid Reduction (APHA *et al.*, 1971)	Not determined	Not determined	Not determined
Potomac Estuary	US EPA 91971)	US EPA (1971)	US EPA (1971)	US EPA (1971)	US EPA (1971)
Redstone	Ascorbic Acid Method (APHA *et al.*, 1971)	Persulfate digestion, followed by Ascorbic Acid Method (APHA *et al.*, 1971)	Phenate Method (APHA *et al.*, 1971)	-	-
Sallie	"As outlined in APHA *et al.*, 1971"				
Sammamish	Molybdate Complexing Reaction (Strickland and Parsons, 1968)		-	Cadmium-Copper Column (Strickland and Parsons, 1968)	-

Table 11a (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - PHOSPHORUS AND NITROGEN CONCENTRATIONS[a]

Water Body	Dissolved Phosphorus	Total Phosphorus	Ammonia	Nitrate	Nitrite
Shagawa	Murphy-Riley Ascorbic Acid Method (US EPA, 1971)	Persulfate digestion, followed by Murphy-Riley Ascorbic Acid Method (US EPA, 1971)	Automated Indophenol Blue Method (US EPA, 1971)	Automated Cadmium Reduction followed by Diazotization (US EPA, 1971)	
Stewart	Ascorbic Acid Method (APHA *et al.*, 1971)	Persulfate digestion, followed by Ascorbic Acid Method (APHA *et al.*, 1971)	Phenate Method (APHA *et al.*, 1971)	-	-
Tahoe	-	-	-	-	-
East Twin	Phosphomolybdate/ Ascorbic Acid Reduction (APHA *et al.*, 1971)	Persulfate Sulfuric Acid digestion, followed by Phosphomolybdate/Ascorbic Acid Reduction (APHA *et al.*, 1971)	Direct Nesslerization (APHA *et al.*, 1971)	Cadmium Reduction (APHA *et al.*, 1971)	-
West Twin	Phosphomolybdate/ Ascorbic Acid Reduction (APHA *et al.*, 1971)	Persulfate Sulfuric Acid digestion, followed by Phosphomolybdate/Ascorbic Acid Reduction (APHA *et al.*, 1971)	Direct Nesslerization (APHA *et al.*, 1971)	Cadmium Reduction (APHA *et al.*, 1971)	-
Twin Valley	Ascorbic Acid Method (APHA *et al.*, 1971)	Persulfate digestion, followed by Ascorbic Acid Method (APHA *et al.*, 1971)	Phenate Method (APHA *et al.*, 1971)	-	-

Table 11a (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - PHOSPHORUS AND NITROGEN CONCENTRATIONS[a]

Water Body	Dissolved Phosphorus	Total Phosphorus	Ammonia	Nitrate	Nitrite
Virginia	Ascorbic Acid Method (APHA *et al.*, 1971)	Persulfate digestion, followed by Ascorbic Acid Method (APHA *et al.*, 1971)	Phenate Method (APHA *et al.*, 1971)	-	-
Waldo	US EPA (1973d)	US EPA (1973d)	US EPA (1973d)	US EPA (1973d)	US EPA (1973d)
Washington	(Note: Many different methods have been used over the years by different investigators. The methods reported here are those of more recent years' studies (Edmondson, 1975b))				
	Phosphomolybdate/ Stannous Chloride Reduction (APHA *et al.*, 1971)	Perchloric Acid digestion, followed by Phosphomolybdate/ Stannous Chloride Reduction (APHA *et al.*, 1971)	Direct Nesslerization (APHA *et al.*, 1971)	"Strychnidine" Method until August, 1967, then Brucine Method (APHA *et al.*, 1971)	-
Weir	US EPA (1971)	US EPA (1971)	Automated Alkaline Phenol Procedure (US EPA, 1971)	Automated Hydrazine Reduction Procedure, Henriksen (1965)	Not determined
Wingra	Murphy and Riley Method (1962)	Persulfate digestion; followed by Murphy and Riley Method (1962)	Alkaline phenol procedure adopted for AutoAnalyzer	Initially Hydrazine Reduction Procedure. Later the Brucine Method of Kahn & Brezenski (1967)	Not determined

[a]As indicated by the US OECD investigators.

Dash (-) indicates no data available.

Table 11b. ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - TRANSPARENCY, PRIMARY PRODUCTIVITY AND CHLOROPHYLL *a* AND DISSOLVED OXYGEN CONCENTRATIONS[a]

Water Body	Water Transparency	Dissolved Oxygen	Chlorophyll *a*	Primary Productivity
Blackhawk	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Brownie	-	-	-	-
Calhoun	-	-	-	-
Camelot-Sherwood Complex	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Canadarago	30 cm white Secchi disc	Weston and Stack D.O. Meter; some surveys made using Winkler Method with Azide Modification (APHA *et al.*, 1971)	Strickland and Parsons (1965) See Hetling *et al.* (1975) for variations between 1968 and subsequent determinations	Method developed by principal investigators (see Hetling *et al.*, 1975 for details)
Cayuga	-	-	-	Not determined
Cedar	-	-	-	Not determined
Dogfish	Secchi disc	-	Strickland and Parsons (1968) until May, 1972; Turner Fluorometer after May, 1972	Not determined
Dutch Hollow	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined

Table 11b (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - TRANSPARENCY, PRIMARY PRODUCTIVITY AND CHLOROPHYLL *a* AND DISSOLVED OXYGEN CONCENTRATIONS[a]

Water Body	Water Transparency	Dissolved Oxygen	Chlorophyll *a*	Primary Productivity
George	-	-	Not determined	^{14}C uptake (Steeman-Nielsen, 1952)
Harriet	-	-	-	Not determined
Isles	-	-	-	Not determined
Kerr Reservoir	8 inch diameter White Secchi disc	Hydrolab Surveyor & Azide Modification of Winkler Method	Turner Fluorometer	Oxygen Production under standard laboratory conditions (i.e., 24°C, 400 foot candles)
Lamb	Secchi disc	-	Strickland and Parsons (1968) until May, 1972; Turner Fluorometer after May, 1972	Not determined
Meander	Secchi disc	-	Strickland and Parsons (1968) until May, 1972; Turner Fluorometer after May, 1972	Not determined
Mendota	Analytical procedures outlined in Lee (1966)			Not determined
Michigan	Analytical procedures outlined in Rousar (1973)			Not determined
Minnetonka	Secchi disc and attenuation coefficients	-	Strickland and Parsons (1968)	-

Table 11b (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - TRANSPARENCY, PRIMARY PRODUCTIVITY AND CHLOROPHYLL a AND DISSOLVED OXYGEN CONCENTRATIONS[a]

Water Body	Water Transparency	Dissolved Oxygen	Chlorophyll a	Primary Productivity
Potomac Estuary	Secchi disc	Winkler Method; Azide Modification (APHA et al., 1971)	90% Acetone extraction	Not determined
Redstone	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Sallie	"As outlined in APHA et al., 1971"			Not determined
Sammamish	Secchi disc	Winkler Method; Azide Modification (APHA et al., 1971)	90% Acetone extraction (Strickland and Parsons, 1968)	^{14}C uptake (Strickland and Parsons, 1968)
Shagawa	Secchi disc	Winkler Method; Azide Modification (EPA, 1971)	90% Acetone extraction (UNESCO, 1966)	Oxygen production; light and dark bottle procedure
Stewart	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Tahoe	-	-	-	-
East Twin	20 cm dia. Secchi disc; alternating black & white quadrants	-	Strickland and Parsons (1968), with trichromatic equations (APHA et al., 1971)	pH method in light and dark bottles after 4 hours of incubation
West Twin	20 cm dia. Secchi disc; alternating black & white quadrants	-	Strickland and Parsons (1968), with trichromatic equations (APHA et al., 1971)	pH method in light and dark bottles after 4 hours of incubation

Table 11b (continued). ANALYTICAL PROCEDURES FOR MAJOR RESPONSE PARAMETERS EXAMINED IN US OECD EUTROPHICATION STUDY - TRANSPARENCY, PRIMARY PRODUCTIVITY AND CHLOROPHYLL *a* AND DISSOLVED OXYGEN CONCENTRATIONS[a]

Water Body	Water Transparency	Dissolved Oxygen	Chlorophyll *a*	Primary Productivity
Twin Valley	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Virginia	Secchi disc	YSI Model 54 D.O. Meter	Strickland and Parsons (1965)	Not determined
Waldo	20 cm white Secchi disc	-	"Strickland and Parsons"	^{14}C uptake
Washington	(Note: Many different methods have been used over the years by different investigators. The methods reported here are those of more recent years' studies (Edmondson, 1975b).			
	Secchi disc	-	Strickland and Parsons (1968) Prior to 1968, used acetone extraction and Klett colorimeter	Oxygen production in light and dark bottles. ^{14}C uptake done for several years
Weir	Secchi disc	-	Trichromatic Method (US EPA, 1973d)	^{14}C uptake (APHA *et al.*, 1971)
Wingra	Secchi disc	YSI D.O. Meter	Not determined	See Huff *et al.* (1972)

[a]As indicated by the US OECD investigators.

Dash (-) indicates no data available.

Table 12. SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Blackhawk, Camelot-Sherwood, Cox Hollow, Dutch Hollow, Redstone, Stewart, Twin Valley and Virginia	A) <u>Phosphorus Loading</u>: 1) Base Flow 2) Woodland 3) Rural Runoff 4) Urban Runoff 5) Manured Lands 6) Precipitation 7) Dry Fallout 8) Domestic Wastewaters 9) Septic Tanks 10) Drained Marshes 11) Groundwater	-Phosphorus loadings estimated from watershed land usage phosphorus export coefficients derived for the Lake Mendota (Wisconsin) watershed and presented in Sonzogni and Lee (1974).
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loadings. Watershed nitrogen export coefficients were used to calculate the nitrogen loadings.
Brownie, Calhoun, Cedar, Harriet and Isles	A) <u>Phosphorus Loading</u>: 1) Waste Discharges (includes city water and air conditioning water) 2) Land Runoff (via storm drain and direct) 3) Estimated Precipitation 4) Estimated Groundwater Input	-No information available.
	B) <u>Nitrogen Loading</u>:	-Not Determined.

Table 12(continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Canadarago	A) <u>Phosphorus Loading:</u>	
	1) Wastewater Discharges	-Estimates were made by direct measurement of the primary wastewater treatment plant to Ocquionos Creek (one of major tributaries to lake), and the difference between upstream and and downstream samples from Ocquionos Creek, and calculations from published per capita contributions.
	2) Septic Tanks	-Estimate made by calculations involving total population of lakeside residences, lakeside residence population having septic tank failures, average residence time of lakeshore facilities and per capita phosphorus input value of 2.9 g P/capita/day. It was assumed any phosphorus entering a septic tank leaching field was retained in the field, unless the tank discharged directly into the lake.
	3) Gaged Tributaries	-Estimated as product of measured daily flows and phosphorus concentrations.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Canadarago (continued)	4) Non-gaged Tributaries	-Assumed runoff for non-gaged area was equal to the average of the area drained by the gaged tributaries, not counting the wastewater treatment plant effluents.
	5) Rainfall and Dry Fallout	-Estimated from literature values; mainly Weibel (1969).
	6) Groundwater	-Considered negligible.
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loadings. For the septic tank nitrogen loadings, 10.3 g N/capita/day was used in the calculations. It was assumed that no nitrogen was retained in the septic tank leaching fields; therefore, it was assumed the entire lakeshore population with septic tanks contributed nitrogen to the lake. Nitrogen fixation was not considered in the nitrogen loading estimates.
Cayuga	A) <u>Phosphorus Loading</u>:	
	1) Waste Discharge	-Determined using estimates of per capita discharge of phosphorus to tributaries and phosphorus in waste discharged directly to lake.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Cayuga (continued)		
	2) *Land Runoff*	-Estimated per capita discharge of phosphorus to tributaries minus phosphorus in waste discharged directly to lake.
	3) Precipitation	-Phosphorus in precipitation monitored in one year study.
	4) Groundwater	-Information not available.
	NOTE: 1) Total phosphorus input and molybdate reactive (unfiltered) phosphorus input taken from Likens (1972b; 1974a; 1974b). 2) Phosphorus in precipitation and in 25 tributaries (draining 78% of watershed) was monitored in a one year study. 3) "Biologically reactive phosphorus" determined using nutrient export coefficients; forest = 8.3 $mg/m^2/yr$; agricultural/rural = 13.2 $mg/m^2/yr$; urban = 100 $mg/m^2/yr$.	
	B) Nitrogen Loading:	
		-Same general methods as for phosphorus loadings.
		-4.44 kg N/yr used as per capita N discharge. (Olsson, Kargren and Tullander, 1968).
		-Sewage treatment efficiency (all types of disposal systems) of 50 percent for N removal was assumed.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Dogfish, Lamb and Meander	A) Phosphorus Loadings:	
	1) Atmosphere (wet and dry)	-Determined by measurement of samples of water collectors placed throughout drainage basin. Snow samples also analyzed.
	2) Surface Flow (sheet flow + flow through soils)	-Measured at two-week intervals during April-October.
	3) Tributary Flow	-Measured at two-week intervals during April-October. Tributaries monitored by grab sample, and flows determined manually on day of sampling.
	4) Groundwater	-Assumed zero.
	Details of 1972 nutrient budgets available in Wright (1974) and Bradbury *et al*. (1974)	
	B) Nitrogen Loading:	-Not determined.
George	A) Phosphorus Loading:	
	1) Runoff 2) Precipitation 3) Sewage Plant Effluents 4) Septic Tank Effluents 5) Lawn Fertilizer	-Taken from Gibble (1974). (Precipitation based on "normal precipitation of basin").
	B) Nitrogen Loading:	-Not Determined

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Kerr Reservoir	A) <u>Phosphorus Loading</u>:	
	1) Point Sources	-Virginia data assembled from tabulation prepared by Hayes, Seay and Mattern for the Roanoke River Basin Study and provided by the Wilmington District, US Army Corps of Engineers. North Carolina data is from Division of Environmental Management, Department of Economic and Natural Resources.
	2) Gaged Tributary Sources	-Information not available.
	3) Non-gaged Tributary Sources	-Equal to total discharge minus gaged stream discharge. Phosphorus and nitrogen concentration estimates from five non-polluted feeder streams were applied to the volume to obtain input from non-gaged sources.
	4) Rainfall	-Taken from nutrient coefficient data of Uttormark *et al.*, (1974) and Gambell and Fisher (1966). Also, total phosphorus was determined on rainfall samples collected at Chapel Hill, North Carolina on April 13 and April 25, 1972.
	5) Groundwater Seepage	-Considered insignificant.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Kerr Reservoir (continued)	B) Nitrogen Loading:	-Same sources and methods as for phosphorus loadings. In addition, dry fallout and nitrogen fixation loadings considered insignificant.
Mendota	A) Phosphorus Loading: 1) Wastewater Discharges 2) Urban Runoff 3) Rural Runoff 4) Precipitation 5) Dry Fallout 6) Groundwater Seepage 7) Base Flow 8) Marsh Drainage	All nutrient loading data taken from Sonzogni and Lee (1974).
	B) Nitrogen Loading:	-Same sources and methods as for phosphorus loadings. In addition, nitrogen fixation was included in the nitrogen loading estimate.
Michigan Open Waters	A) Phosphorus Loading:	-1971 phosphorus loadings were taken from Lee (1974a) and included phosphorus loadings from: 1) direct wastewater, 2) indirect wastewater, 3) erosion and other diffuse sources, 4) combined sewer overflow, and 5) precipitation and dry fall-out onto lake surface.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Michigan (Open Waters) (continued)		-1974 phosphorus loadings were taken from Lee (1974a).
	B) <u>Nitrogen Loading</u>:	-Taken from Bartsch (1968)
Michigan		
Nearshore Waters	--Nutrient Loadings Not Determined	
Offshore Waters	--Information Not Available	
Lower Lake Minnetonka	A) <u>Phosphorus Loading</u>: 1) Sewage Effluents 2) Tributary Streams 3) Overland Runoff 4) Rainfall on Lake 5) Septic Tank Drainage	-All nutrient loading data taken from compilations made by Harza Engineering Company ("A Program For Preserving The Quality Of Lake Minnetonka"). State of Minnesota Pollution Control Agency, Minneapolis, Minnesota. 1971. (Megard, 1975). -Overland runoff was estimated as 130 $lbs/mi^2/yr$ for rural runoff and 510 $lbs/mi^2/yr$ for urban runoff. -Phosphorus concentration in rainfall assumed to be 20 mg/m^3.
	B) <u>Nitrogen Loading</u>:	-Not Determined.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Potomac Estuary	A) Phosphorus Loading:	
	1) Upper Basin Runoff	-Based on two years of weekly sampling of upper basin runoff.
	(Note: Upper basin runoff includes both land runoff and waste-water discharges in upper basin)	
	2) Estuarine Wastewater Discharges	-Based on two years of weekly samplings of point sources.
	3) Precipitation 4) Groundwater	-Considered insignificant. Dry fallout not considered in phosphorus loading estimate.
	B) Nitrogen Loading:	-Same sources and methods as for phosphorus loadings. Nitrogen fixation and dry fall-out not considered in nitrogen loading estimate.
Sallie	A) Phosphorus Loading:	
	1) Waste Discharge	-Waste discharged from City of Detroit Lakes into Pelican River which discharges into lake. Concentrations of phosphorus in ditch to river was monitored and converted to weight.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Sallie (continued)	2) Land Runoff	-Estimated as total in Pelican River minus waste load total in other surface inlets.
	3) Precipitation	-Phosphorus concentration in precipitation was monitored and converted to weight as product of lake area and total precipitation.
	4) Groundwater	-Collected with investigator-designed sampler as it entered lake. Phosphorus weight was calculated for discharge increase over surface inflow.
	B) Nitrogen Loading:	-Same sources and methods for phosphorus loadings.
Sammamish	A) Phosphorus Loading:	
	1) Waste Discharge	-Several independent methods.
	2) Land Runoff	-Equal to total phosphorus loading plus precipitation phosphorus loading.
		-Total phosphorus loading equal to sum of measurement of 13 streams and pipes entering lake plus waste contributions by several independent methods.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Sammamish (continued)	3) Precipitation	-Atmospheric phosphorus input to lake surface determined from limited rainwater analysis during 1971 water year.
	4) Groundwater	-Determined as insignificant because water balance was explainable from consideration of surface inputs and outputs.
	B) <u>Nitrogen Loading</u>:	-Same sources and methods for phosphorus. In addition, dry fallout nitrogen input was not considered in nutrient loading estimate. Nitrogen fixation was considered insignificant.
Shagawa	A) <u>Phosphorus Loading</u>:	
	1) Waste Discharges	-In 1971 and earlier years, waste discharges determined from single daily grab samples and some four and six hour-nonweighted composites obtained. In 1972, waste discharges computed phosphorus concentrations in the wastewater obtained from 24 hour flow-weighted composite samples. Loadings were the product of composite concentrations and the total daily flows.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Shagawa (continued)	2) Land Runoff to Tributaries	-Weekly, nonflow-weighted phosphorus concentrations were integrated to obtain daily values for creeks. Daily loads were product of concentration and daily flow. Prior to 1972, monthly loading was product of monthly mean phosphorus concentration and total stream flow for month. Non-gaged tributaries estimated as ratio of non-gaged to gaged area, and multiplying the loading by the factor.
	3) Precipitation	-Estimated using average phosphorus concentration collected at Ely, Minnesota, and multiplying by the monthly precipitation falling on the lake.
	4) Other (= direct runoff + excess drinking water)	-An average load/unit area/month was calculated based on the load/unit area/month for the gaged basins.
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loadings. Nitrogen inputs from wastewater treatment plants were calculated in a manner similar to that used to determine the phosphorus loadings.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Tahoe	A) <u>Phosphorus Loading</u>:	
	NOTE: According to state and federal regulations no wastewater is supposed to be discharged within the drainage basin.	
	1) Land Runoff (1969 data)	-Total monthly discharge of nine major tributaries calculated from daily USGS flow measurements. Total monthly discharge of other 54 creeks and tributaries estimated as in McGauhey *et al.*(1963). Phosphorus concentration data collected on nine major tributaries by the Tahoe Research Group of the Univ. of California at Davis, the California-Nevada Federal Joint Water Quality Investigation, Lake Tahoe Area Council and the Water Resources Information Series of the State of Nevada. Total phosphorus mass calculated as product of total flow and mean concentration.
	2) Precipitation	-Only traces of phosphorus were assumed to be present in rainfall.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Tahoe (continued)	3) Groundwater	-Assumed insignificant input.
	B) Nitrogen Loading:	-The same sources and methods as for phosphorus loading. In addition, the average NH_4^+-N and NO_3^--N were measured in the precipitation to estimate the total nitrogen input from rainfall.
Twin Lakes	A) Phosphorus Loading:	
	1) Waste Discharges	-Assumed zero.
	2) Land Runoff ("sheet" runoff)	-Computed from lake level increases, as recorded by limnographs, in excess of that from direct precipitation and stream inflows.
	3) Precipitation	-Measured with a recording Leupold-Stevens type Q6 weighing bucket located at West Twin Lake. Rain and snow samples (which included dry fallout) were collected at Kent State University, Kent, Ohio, for nutrient analysis.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Twin Lakes (continued)	4) Groundwater	-Twenty-eight shallow wells were installed around lake perimeter and a flow net constructed. Specific discharge determined from hydraulic gradient and field measurement of permeability. Wells were sampled monthly for nutrient content.
	5) Surface Streams	-Measured daily or continuously depending on station, mainly with either 90° V notch weir and stilling well or bucket or culvert discharge and current meter. Dollar Lake Stream Station was measured daily with either culvert discharge and bucket or 60° or 90° V notch weir and stilling basin or well.
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loadings. Nitrogen fixation was not included in the nitrogen loading estimates.
Waldo	A) <u>Phosphorus Loading</u>: NOTE: Dry fallout was not considered in phosphorus loading estimate. Marsh drainage considered insignificant.	-Estimated using four indirect methods as follows: 1) Using information from Vollenweider (1975a) assume phosphorus loading = three times measured lake concentration = three times mean outflow concentration;

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Waldo (continued)		2) Using watershed phosphorus export coefficients derived for undisturbed forest land in Upper Klamath Lake, Oregon (Miller, unpublished data in Powers *et al.*, 1975). 3) Using average precipitation data for the lake and snow analyses of Malueg *et al.* (1972) and assuming a) all precipitation into watershed eventually enters lake, or b) only the precipitation equal to measured outflow plus estimated evaporation actually enters lake; and 4) Using total phosphorus soil export factors of Vollenweider and Dillon (1974), and assuming remainder of loading is direct precipitation onto the lake surface. The mean of the four estimated values was reported as the annual phosphorus loading.
	B) Nitrogen Loading: (NOTE: Dry fallout was not considered in nitrogen loading estimate; marsh drainage and nitrogen fixation considered insignificant)	Estimated using methods 2, 3a and 3b above. (Method 1 not used because estimates of nitrogen retention in lake unknown. Method 4 not used because of lack of information on soil loading of nitrogen to lake).

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Washington	A) <u>Phosphorus Loading</u>:	
	(NOTE: Several sampling regimes and analytical methodologies were used by different investigators over the years, making a concise summary difficult)	
	1957	-All sewage plants and many tributaries to the lake sampled twice per week by the Seattle Engineering Department. Nutrient concentrations, including total phosphorus, phosphate and particulate phosphorus were determined using methods listed in APHA <u>et al</u>. (1971), and earlier editions.
	1964	-METRO analyzed fewer tributaries (10) for fewer parameters (i.e., total phosphorus, Kjeldahl nitrogen and nitrate plus nitrite nitrogen) approximately weekly.
	1970's	-The two major inlets and one minor inlet sampled biweekly by the US OECD investigator for total phosphorus and phosphate (in 1957, these two major inlets supplied 86% of total phosphorus loading. The total phosphorus loading is approximated by proportion).

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Washington (continued)	Two sources of water flow data were used. The major source was gage data published by USGS. In 1957, the USGS was gaging the two major inlets + two smaller inlets. The rest of the tributaries were determined by proportion with the watershed area. A hydrological model was developed later for METRO and used until 1972 to estimate the Sammamish input. Since 1972, a regression equation that relates total Sammamish flow to stations that are gaged in the watershed has been used to determine the water flow.	
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loading. In 1957, the Seattle Engineering Department analyzed the input water for "several nitrogen components". In 1964, METRO analyzed the samples for Kjeldahl nitrogen and nitrate plus nitrite nitrogen. In 1970's the US OECD investigator has been analyzing for nitrate, nitrite, ammonia and Kjeldahl nitrogen. The sources of flow data are the same as for the phosphorus loading.
Weir	A) <u>Phosphorus Loading</u>:	
	1) Rainfall	-Taken from Brezonik <u>et al</u>. (1969) for rainfall at Gainesville, 60 miles north of lake.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Weir (continued)	2) Urban	-Urban runoff values taken from Weibel (1969) and represents averages for residential-light commercial areas found in study area.
	3) Pasture 4) Forest	-Pasture and forest runoff values taken from Uttormark *et al*. (1974). In order to account for low nutrient binding capacity of sandy acid soils in study area, the "average" and "high" areal yield rates of Uttormark *et al*. (1974) were averaged for these two land-use classifications.
	5) Agriculture	-Taken from estimates of Brezonik and Shannon (1971) based on the average fertilizer composition and application rate to citrus groves.
	6) Septic Tank	-Estimated using methods of Brezonik and Shannon (1971). Average septic tank daily effluent flow of 475 l, with total phosphorus concentration of 8 mg/l, was assumed. For lakeshore houses, it was assumed 10 percent of the phosphorus was transported to the lake. For non-lakeshore houses, it was assumed one percent of the phosphorus was transported to the lake.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Weir (continued)	7) Wetlands	-Net phosphorus contribution assumed zero.
	B) Nitrogen Loading:	-Same sources and methods as above. For the septic tank nitrogen loadings, a total nitrogen concentration in the septic tank effluent of 35 mg/l was assumed (Brezonik and Shannon, 1971). It was assumed 25 percent of the lakeshore homes nitrogen loading and 10 percent of the non-lakeshore homes nitrogen loading were transported to the lake.
Wingra	A) Phosphorus Loading:	
	1) Precipitation	-Rain and snow were collected in open bucket type containers which were put out when precipitation seemed imminent.
	2) Dry Fallout	-Estimated by exposing container to atmosphere for several days. During winter, bulk precipitation was measured rather than dry fallout.
	3) Springflow	-Monitored continuously by USGS where possible. Samples collected every two weeks for phosphorus determinations.

Table 12 (continued). SUMMARY OF METHODS USED TO CALCULATE NUTRIENT LOADINGS FOR US OECD WATER BODIES

Water Body	Nutrient Sources Considered by US OECD Investigator in Nutrient Loading Estimates	General Methodology Indicated by Investigator to Determine the Nutrient Loading to Water Body
Wingra (continued)	4) Urban Runoff	-Determined by measurements taken from the Manitou Way Basin, especially during storm periods (Kluesener, 1972).
	5) Groundwater	-Considered insignificant (Kluesener, 1972).
	6) Marsh	-Assumed marsh input loads roughly equal to marsh output loads. Therefore, marsh net phosphorus contribution is zero.
	B) <u>Nitrogen Loading</u>:	-Same sources and methods as for phosphorus loadings.

results are far from complete. While all investigators reported the nutrient sources they considered in their nutrient budget estimates, in some instances sufficient detail was not given as to exactly how the nutrient loadings were estimated. For example, if watershed land use nutrient export coefficients were used, what was the distribution of land use types in the watershed? How was the percentage of different watershed land use types calculated? How were the export coefficients calculated or estimated? If nutrient inputs were measured directly, what analytical methods were used? What nutrients were measured? What was the sampling frequency? How were the tributaries sampled? How many of the tributaries were sampled? What percent of the tributary area was sampled? These are major questions that must be answered before the usefulness of US OECD eutrophication study data, as applied in the Vollenweider phosphorus loading diagrams and other loading diagrams, can be fully determined.

The major nutrient input sources, according to most US OECD investigators, were wastewater discharges, land runoff and precipitation. Most US OECD investigators also considered groundwater inputs in their nutrient budget calculations, although these inputs were generally considered insignificant nutrient sources. A summary of the various nutrient sources considered in the nutrient loading calculations, as indicated by the US OECD investigators, is presented in Table 13.

METHODS FOR EVALUATION OF ESTIMATES OF US OECD WATER BODY NUTRIENT LOADINGS

Suffiently detailed information concerning the methodology used in estimating the nutrient budgets for the US OECD eutrophication study water bodies was not available in most cases. As a result, several independent methods were employed by these reviewers in an attempt to check the reasonableness of the nutrient loadings reported by the US OECD investigators. These methods include the use of several relationships developed by Vollenweider (which relate phosphorus loadings to mean water body phosphorus concentrations) and the use of watershed nutrient export coefficients and land usage patterns within the watershed of a water body to predict phosphorus and nitrogen loadings. These methods were not developed as an absolute guide for evaluating the accuracy of the US OECD investigators' nutrient loadings, but rather are meant to serve as a basis for checking on the reasonableness of these loadings, with the goal of detecting any possible major errors or unusual water body situations. An identification key for the US OECD water bodies is presented in Table 14. This key will be used in all subsequent figures to identify the US OECD water bodies.

Table 13 SUMMARY OF NUTRIENT SOURCES CONSIDERED IN US OECD WATER BODY NUTRIENT LOADING ESTIMATES

Water Body	Trophic State[a]	Waste Water Discharges	Urban and/or Rural Land Runoff	Precipitation onto Water Body Surface	Dry Fallout onto Water Body Surface	Ground Water Seepage	Woodland Runoff	Marsh Drainage	Nitrogen Fixation
Blackhawk[b]	E	+	+	+	+	*	+	+	-
Brownie	E	+	+	+	-	+	+	+	**
Calhoun	E	+	+	+	-	+	+	+	**
Camelot-Sherwood Complex[b]	E	+	+	+	+	+	+	+	-
Canadarago	E	+	+	+	+	0	0	0	-
Cayuga	M	+	+	+	+	*	+	-	-
Cedar	E	+	+	+	-	+	+	+	**
Cox Hollow[b]	E	+	+	+	+	+	+	+	-
Dogfish	0	+	+	+	+	0	+	0	**
Dutch Hollow[b]	E	+	+	+	+	+	+	+	-
George	0-M	+	+	+	+	0	+	-	-
Harriet	E	+	+	+	-	+	+	+	**
Isles	E	+	+	+	-	+	+	+	**
Kerr Reservoir	E-M	+	+	+	0	0	+	+	0
Lamb	0	+	+	+	+	0	+	0	**

Table 13 (Continued). SUMMARY OF NUTRIENT SOURCES CONSIDERED IN US OECD WATER BODY NUTRIENT LOADING ESTIMATES

Water Body	Trophic State[a]	Waste Water Discharges	Urban and/or Rural Land Runoff	Precipitation onto Water Body Surface	Dry Fallout onto Water Body Surface	Ground Water Seepage	Woodland Runoff	Marsh Drainage	Nitrogen Fixation
Meander	0	+	+	+	+	0	+	0	**
Mendota	E	+	+	+	+	+	0	0	+
Michigan	0-M	+	+	+	+	+	+	+	-
Minnetonka	E→M	+	+	+	-	-	-	+	**
Potomac Estuary	U-E	+	+	0	-	0	+	-	-
Redstone[b]	E	+	+	+	+	+	+	+	-
Sallie	E	+	+	+	-	+	+	+	-
Sammamish	M	+	+	+	-	0	+	+	-
Shagawa	E	+	+	+	+	-	+	0	-
Stewart[b]	E	+	+	+	+	+	+	+	-
Tahoe	U-0	+	+	+	-	0	+	+	-
Twin Lakes	E	+	+	+	*	+	+	0	*
Twin Valley[b]	E	+	+	+	+	+	+	+	-
Virginia[b]	E	+	+	+	+	+	+	+	-

Table 13 (Continued). SUMMARY OF NUTRIENT SOURCES CONSIDERED IN US OECD WATER BODY NUTRIENT LOADING ESTIMATES

Water Body	Trophic State[a]	Waste Water Discharges	Urban and/or Rural Land Runoff	Precipitation onto Water Body Surface	Dry Fallout onto Water Body Surface	Ground Water Seepage	Woodland Runoff	Marsh Drainage	Nitrogen Fixation
Waldo	U-O	+	+	+	-	+	+	0	-
Washington (1974)	M	+	+	-	-	+	+	+	-
Weir	M	+	+	+	-	+	+	0	-
Wingra	E	+	+	+	+	+	+	+	-

EXPLANATION:

+ = considered in nutrient budget calculations
- = not considered in nutrient budget calculations
0 = considered to be insignificant in nutrient budget
* = considered in nutrient budget calculation, but significance unknown
** = nitrogen budget not calculated

[a]Investigator indicated trophic state:
E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

[b]Nutrient budget calculated from watershed land use nutrient export coefficients.

Table 14. IDENTIFICATION KEY FOR US OECD WATER BODIES

Water Body	Identification Number	Investigator-Indicated Trophic Status	Location
Blackhawk	1	Eutrophic	Wisconsin
Brownie	2	Eutrophic	Minnesota
Calhoun	3	Eutrophic	Minnesota
Camelot-Sherwood	4	Eutrophic	Wisconsin
Canadarago			New York
-1968	5-A	Eutrophic	
-1969	5-B		
Cayuga			New York
-1972	6-A	Mesotrophic	
-1973	6-B		
Cedar	7	Eutrophic	Minnesota
Cox Hollow	8	Eutrophic	Wisconsin
Dogfish			
-1971	9	Oligotrophic	Minnesota
-1972	10	Oligotrophic	
Dutch Hollow	11	Eutrophic	Wisconsin
George	12	Oligotrophic-Mesotrophic	New York
Harriet	13	Eutrophic	Minnesota
Isles	14	Eutrophic	Minnesota
Kerr Reservoir		Eutrophic-Mesotrophic	North Carolina, Virginia
Whole Reservoir	15		
-Roanoke Arm	16		
-Nutbush Arm	17		
Lamb			Minnesota
-1971	18	Oligotrophic	
-1972	19	Oligotrophic	

Table 14 (Continued) IDENTIFICATION KEY FOR US OECD WATER BODIES

Water Body	Identification Number	Investigator-Indicated Trophic Status	Location
Meander			Minnesota
-1971	20	Oligotrophic	
-1972	21	Oligotrophic	
Mendota	22	Eutrophic (changing)	Wisconsin
Michigan (open waters)			Michigan, Wisconsin
-1971 } τ_{ω} = 30 yrs	23-A	Oligotrophic	
-1974	24-A	Oligotrophic	
-1971 } τ_{ω} = 100 yrs	23-B		
-1974	24-B		
Michigan (nearshore waters)			
-1971	23-C		
-1974	24-C		
Lower Lake Minnetonka			Minnesota
-1969	25	Eutrophic	
-1973	26	Eutrophic (changing)	
Potomac Estuary			Maryland, Virginia
Whole Estuary	27	Ultra-Eutrophic	
-Upper Reach	28		
-Middle Reach	29		
-Lower Reach	30		
Redstone	31	Eutrophic	Wisconsin
Sallie	32	Eutrophic	Minnesota
Sammamish	33	Mesotrophic	Washington
Shagawa	34	Eutrophic	Minnesota
Stewart	35	Eutrophic	Wisconsin
Tahoe	36	Ultra-Oligotrophic	California, Nevada

Table 14 (Continued) IDENTIFICATION KEY FOR US OECD WATER BODIES

Water Body	Identification Number	Investigator-Indicated Trophic Status	Location
Twin Lakes	37	Eutrophic (changing)	Ohio
East Twin Lake	38		
-1972	39	Eutrophic	
-1973	40	Eutrophic	
-1974	41	Eutrophic	
West Twin Lake	42		
-1972	43	Eutrophic	
-1973	44	Eutrophic	
-1974	45	Eutrophic	
Twin Valley	46	Eutrophic	Wisconsin
Virginia	47	Eutrophic	Wisconsin
Waldo	48	Ultra-Oligotrophic	Oregon
Washington			Washington
-1957	49	Eutrophic	
-1964	50	Eutrophic	
-1971	51	Mesotrophic	
-1974	52	Mesotrophic	
Weir	53	Mesotrophic	Florida
Wingra	54	Eutrophic	Wisconsin

Vollenweider Mean Phosphorus/Influent Phosphorus And Hydraulic Residence Time Relationship

The first method used by these reviewers to check the reasonableness of the US OECD eutrophication study phosphorus loading estimates involved the use of the relationship between the average influent phosphorus concentration and the mean phosphorus concentration in the water body. Equation 20 may be rearranged as follows:

$$[P]_\infty/(L(P)/q_s) = 1/(1 + \sqrt{\bar{z}/q_s}) \qquad (25)$$

Recalling $L(P)/q_s = \overline{[P]}$ and $\bar{z}/q_s = \tau_\omega$, then Equation 25 becomes

$$[P]_\infty/\overline{[P]} = 1/(1 + \sqrt{\tau_\omega}) \qquad (26)$$

According to Vollenweider (1975b; 1976a), the average influent phosphorus concentrations are generally higher than the mean water body phosphorus concentrations because of the continuous loss of phosphorus to the sediments. In a highly flushed water body (i.e., hydraulic residence time, τ_ω, < 0.5 yr), which would exhibit very little relative sedimentation of phosphorus because of the rapid flow of phosphorus through the water body, the ratio of the mean phosphorus concentration to the influent phosphorus concentration approaches unity. With less rapidly flushed water bodies, there is an increasing involvement of the input phosphorus with the water body metabolism and a resultant deviation of this ratio from unity. This deviation can become positive or negative, depending on whether phosphorus accumulates in the water phase or the sediment phase of the water body. In actuality, the ratio of the water body mean phosphorus concentration to the average influent phosphorus concentration defines the ratio of the residence time of phosphorus to the residence time of water (i.e., $\bar{\tau}_p/\tau_\omega=\pi_r$), though in principle this definition applies to any substance flowing into a water body. It can also be used to check on the phosphorus sedimentation rate (Vollenweider, 1976a). The derivation and implications of the relative phosphorus residence time, π_r, have been discussed in an earlier section of this report (See Equations 13-16).

The reasonableness of the US OECD eutrophication study phosphorus loading estimates can be checked with the use of Equation 26. A water body's influent phosphorus concentration, $\overline{[P]}$, can be calculated as $L(P)/q_s$. The ratio of its mean phosphorus to influent phosphorus concentration, $[P]/\overline{[P]}$, can then be compared to its hydraulic residence time expression, $1/(1 + \sqrt{\tau_\omega})$. The relationship expressed above in Equation 26 can be used as a check on the phosphorus loading estimates since the influent phosphorus concentration is a function of the phosphorus loading. Any major deviations of $[P]/\overline{[P]}$ from $1/(1 + \sqrt{\tau_\omega})$ would make the reported phosphorus loading data suspect. Vollenweider has used this relationship successfully to trace loading errors in the phosphorus budgets for Lakes Constance (Vollenweider, 1975c) and Lunzer See (Vollenweider, 1975d). The use of Equation 26 to check on the accuracy of a water body's phosphorus loading estimate requires that the water body mean phosphorus concentration be accurately known. No equivalent relationship has been derived by Vollenweider for checking nitrogen loading estimates, although a similar approach would likely be applicable.

The relationship expressed in Equation 26 has been applied to the US OECD eutrophication study phosphorus loading estimates. The pertinent data are presented in Table 15. A missing water body identification number indicates that necessary data for the relationship expressed in Equation 26 were not available for a given water body for a particular time period. For example, there is insufficient data for Dogfish Lake-1971 (Identification Number 9). Consequently, it was not included in Table 15. Similar reasoning holds for any missing water body identification numbers in any of the tables in this report. Refer to Table 14 for identification of any water bodies and/or time periods not included in a given table or figure in this report. A plot (Figure 14) has been prepared which graphically illustrates the relationship indicated in Equation 26. The US OECD data, as reported by the US OECD investigators, are also presented in Figure 14. If a data range was reported for a water body, the mean value was used in all calculations. The solid line in Figure 14 signifies a perfect agreement between $[P]/\overline{[P]}$ and $1/(1+\sqrt{\tau_\omega})$. According to the Vollenweider relationship (Equation 26), if the phosphorus loading was overestimated (i.e., the phosphorus loading L(P) is actually smaller than that reported by the US OECD investigator), then the water body would plot below the solid line. Conversely, if the phosphorus loading were underestimated (i.e., the phosphorus loadings are actually higher than those reported by the investigator), the water body would plot above the solid line. The broken lines indicate the degree of possible over- or underestimation of the US OECD investigator-indicated phosphorus loadings relative to that predicted by the hydraulic residence time expression in Equation 26. The "+2x" broken line below the solid line indicates the US OECD investigator-indicated phosphorus loading estimate may have been overestimated (i.e., +) a factor of 2 (i.e., 2x). Conversely, the "-3x" broken line above the solid line indicates the phosphorus loading estimates may have been underestimated (-) by a factor of 3 (3x). The shaded zone between ± 2x indicates the range within which the phosphorus loadings were considered to be reasonable by these reviewers. The basis for the choice of this range of acceptable deviation will be discussed further in a following section.

As can be seen in Figure 14, almost no water bodies fall directly on the solid line. However, many of the water bodies fall within the shaded area between the broken lines representing a ± two-fold possible phosphorus loading estimate error. This indicates the US OECD phosphorus loading estimates generally appear to

Table 15. US OECD DATA FOR VOLLENWEIDER'S MEAN PHOSPHORUS/ INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading (mg P/m^2/yr)[b]	Hydraulic Loading, q_s (m/yr)[c]	Influent Phosphorus Concentration $\overline{[P]}$ (mg/m^3)[d]	Mean Phosphorus Concentration, [P] (mg/m^3)[b]	$\frac{[P]}{\overline{[P]}}$	$\frac{1}{(1+\sqrt{\tau_\omega})}$
Blackhawk (1)[e]	E	2220	9.8	227	50-120	0.22-0.52	0.58
Brownie (2)	E	1180	3.4	347	-	-	0.41
Calhoun (3)	E	860	2.94	292	106[f]	0.36	0.34
Camelot-Sherwood(4)	E	2350-2680	21.4-33.3	70.6-125	30-40	0.24-0.57	0.73-0.77
Canadarago (5)	E	800	12.8	62.4	40-50	0.64-0.80	0.56
Cayuga (6)	M	800	6.3	127	20	0.15	0.25
Cedar (7)	E	350	1.8	189	55[f]	0.29	0.36
Cox Hollow (8)	E	1620-2080	5.4-7.6	213-385	60-100	0.16-0.47	0.54-0.58
Dogfish (10)	O	20	1.14	17.5	10	0.57	0.48
Dutch Hollow (11)	E	950-1010	1.67	569-605	120-400	0.20-0.70	0.43
George (12)	O-M	70	2.25	31.1	8.5	0.27	0.26
Harriet (13)	E	710	3.67	194	62[f]	0.32	0.39
Isles (14)	E	2030	4.5	451	110[f]	0.24	0.56
Kerr Reservoir	E-M						
Roanoke Arm (16)	-	5200	51.5	101	30	0.30	0.69
Nutbush Arm (17)	-	700	1.6	435	30	0.07	0.31

Table 15 (continued). US OECD DATA FOR VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading (mg P/m^2/yr)[b]	Hydraulic Loading, q_s (m/yr)[c]	Influent Phosphorus Concentration $\overline{[P]}$ (mg/m^3)[d]	Mean Phosphorus Concentration, [P] (mg/m^3)[b]	$\frac{[P]}{\overline{[P]}}$	$\frac{1}{(1+\sqrt{\tau_\omega})}$
Lamb (19)	O	30	1.74	17.2	12-13	0.69-0.76	0.40
Meander (21)	O	30	1.85	16.2	9-12	0.56-0.74	0.38
Mendota (22)	E	1200	2.67	450	150	0.33	0.32
Michigan							
Open Waters (23-A)	O	140	2.8	50	13	0.26	0.15
(23-B)		140	0.84	167	13	0.08	0.09
Lower Lake Minnetonka							
1969 (25)	E	500	1.32	379	60	0.16	0.28
1973 (26)	E→M	100(180)[i]	1.32	76(136)	50	0.66(0.37)	0.28
Potomac Estuary	U-E						
Upper Reach (28)	-	85000	120.0	708	300-1200[g]	0.42-1.69	0.45
Middle Reach (29)	-	8000	28.3	282	10-750[g]	0.04-2.66	0.70
Lower Reach (30)	-	1200	8.47	142	30-60[g]	0.21-0.42	0.52
Redstone (31)	E	1440-1680	4.3-6.1	236-390	30-110	0.08-0.47	0.50-0.54
Sallie (32)	E	1500-4200	3.6-5.8	259-1167	350	0.30-1.36	0.43-0.49
Sammamish (33)	M	700	10.0	70	30	0.43	0.47
Shagawa (34)	E	700	7.12	98.2	60	0.61	0.53
Stewart (35)	E	4820-8050	23.8	202-338	40-80	0.12-0.40	0.78

Table 15 (continued). US OECD DATA FOR VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading (mg P/m^2/yr)[b]	Hydraulic Loading, q_s (m/yr)[c]	Influent Phosphorus Concentration $\overline{[P]}$ (mg/m^3)[d]	Mean Phosphorus Concentration, [P] (mg/m^3)[b]	$\frac{[P]}{\overline{[P]}}$	$\frac{1}{(1+\sqrt{\tau_\omega})}$
Tahoe (36)	U-O	50	0.45	112	3	0.03	0.04
East Twin							
1972 (39)	E	700(672)[j]	6.25(7.40)[j]	112 (91)[j]	80 (83)[j]	0.71 (0.91)[j]	0.53 (0.54)[j]
1973 (40)	E	500(472)	5.56(7.19)	89.9(66)	80(78)	0.89(1.81)	0.51(0.54)
1974 (41)	E	700(816)	10.0(9.31)	70(76)	80(77)	1.14(1.01)	0.59(0.58)
West Twin							
1972 (43)	E	400(419)	2.71(0.79)	148(123)	120(122)	0.81(0.99)	0.44(0.47)
1973 (44)	E	300(181)	2.41(0.64)	124(65)	110(107)	0.89(1.65)	0.43(0.45)
1974 (45)	E	300(316)	4.34(1.03)	69.1(75)	100(97)	1.45(1.29)	0.50
Twin Valley (46)	E	1740-2050	7.6-9.5	183-270	60-70	0.22-0.38	0.58-0.61
Virginia (47)	E	1150-1480	0.61-1.89	608-2426	20-150	0.01-0.25	0.37-0.51
Waldo (48)	U-O	17	1.71	9.9	< 5[h]	< 0.5	0.18
Washington							
1957 (49)	E	1200	13.8	87.0	24	0.28	0.39
1964 (50)	E	2300	13.8	167	66	0.40	0.39
1971 (51)	M	430	13.8	31.3	18	0.58	0.39

Table 15 (continued). US OECD DATA FOR VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading (mg P/m²/yr)[b]	Hydraulic Loading, q_s (m/yr)[c]	Influent Phosphorus Concentration $[\overline{P}]$ (mg/m³)[d]	Mean Phosphorus Concentration, [P] (mg/m³)[b]	$\frac{[P]}{[\overline{P}]}$	$\frac{1}{(1+\sqrt{\tau_\omega})}$
Weir (53)	M	140	1.5	93	80	0.86	0.33
Wingra (54)	E	900	6.0	150	70	0.47	0.61

[a]Investigator-indicated trophic states:

E = eutrophic   O = oligotrophic
M = mesotrophic   U = ultra

[b]Based on investigator estimates.

[c]Hydraulic loading, q_s = mean depth, $\bar{z}$/hydraulic residence time, τ_ω.

[d]Influent phosphorus concentration, $[\overline{P}]$ = phosphorus loading, L(P)/hydraulic loading, q_s.

[e]Indicates identification number for Figure 14 (See Table 14).

[f]Summer surface average value.

[g]Summer average value.

[h]August average value.

[i]Data in parentheses represent data received by these reviewers from the principal investigator subsequent to completion of this report. Figure 14 is based on the original data reported by the investigator and does not reflect the altered data. Examination of the data indicated the 1973 phosphorus load was underestimated by approximately two-fold. It is noted the revised loading corresponds to the predicted results in Figure 14.

[j]Data in parentheses represent data received by these reviewers from the principal investigator subsequent to completion of this report. Figure 14 is based on the original data reported by the investigator and does not reflect the changes indicated above. Examination of this subsequent data indicated the phosphorus loads were originally underestimated; however, there were no significant changes in the overall conclusions concerning the Twin Lakes as a result of these altered values.

Dash (-) indicates data not available.

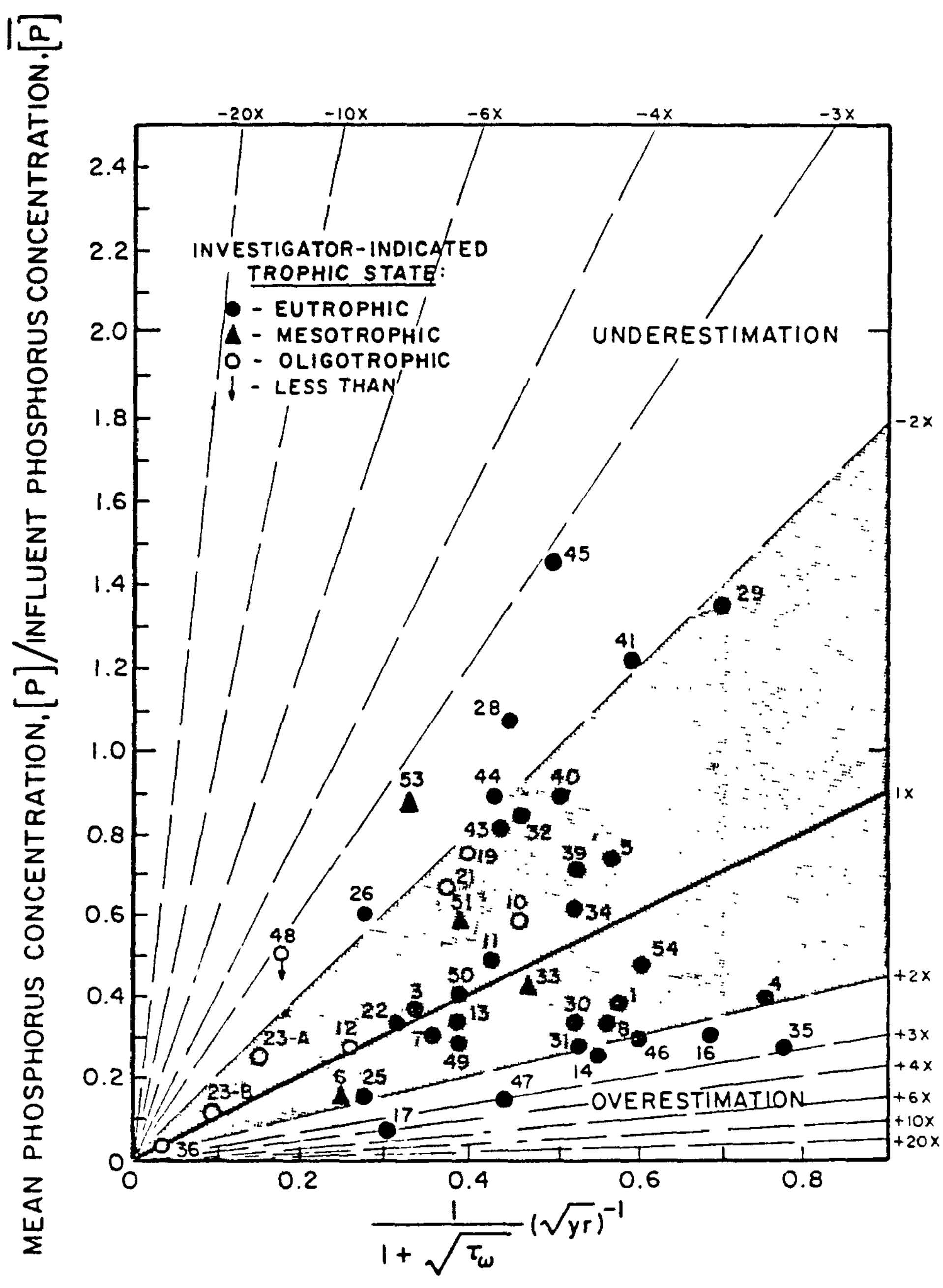


Figure 14. Evaluation of Estimates of US OECD Water Body Nutrient Loadings: Vollenweider Mean Phosphorus/Influent Phosphorus and Hydraulic Residence Time Relationship

be of a reasonable nature, based on the Vollenweider relationship (Equation 26). Considering the multitude of methods used in estimation of the phosphorus loadings (Table 12), this initial agreement between the phosphorus loadings as indicated by the US OECD investigators and the phosphorus loadings as indicated by the Vollenweider relationship (Equation 26) is reassuring and provides some affirmation of the Vollenweider loading diagram approach to establishing the critical phosphorus loading levels and relative trophic conditions of water bodies. Equation 26 will be discussed in greater detail in relation to the Vollenweider phosphorus loading diagrams presented in subsequent sections of this report.

Watershed Land Use Nutrient Export Coefficients

The other principal method used by these reviewers for checking the reasonableness of the phosphorus loading estimates, as well as the nitrogen loading estimates, reported by the US OECD investigators was to compare the reported loadings with those computed using watershed nutrient export coefficients. The nutrient export coefficients used to estimate the nutrient loadings from a given watershed would depend on the land usage pattern within the watershed. Because no relationship equivalent to Equation 26 has been derived for nitrogen loadings, the use of watershed nitrogen export coefficients represents the only independent method available to these reviewers for checking the accuracy of the nitrogen loadings reported by the US OECD investigators.

This procedure involves utilization of the information available on land usage within a lake's or impoundment's watershed and the nutrient coefficients which are applicable to the various land uses within that watershed. For example, a hectare of corn or a suburban subdivision are known to yield a relatively constant amount of aquatic plant nutrients over the annual cycle (see Sonzogni and Lee (1974), for further discussion of this approach). The use of this approach for computing nutrient loadings to a water body requires an accurate estimation of the water body's watershed area and the land usage pattern within the watershed. The US OECD investigators reported watershed land usage in varying degrees, with some investigators producing only sparse watershed land usage data, while others went into great detail concerning land usage within the watershed.

Uttormark *et al.* (1974), based on the results of their extensive survey, have reported there is little justification for the delineation of land usage within direct drainage basins beyond four categories: urban, forest, agriculture and wetlands. Available data are too fragmentary and variable to warrant further subdivision of land usage categories, according to Uttormark *et al.* (1974). The US EPA has taken the same general approach in categorizing watershed land usage types as urban, agriculture, mostly agriculture, forest, mostly forest and mixed (US EPA, 1974c; 1975c). Vollenweider (1977) has recently indicated, based on studies of German watershed land usage, that a distinction between arable land and pastures and meadows may be useful because these two classes of land use types export distinctly different quantities of phosphorus and nitrogen from the watershed. However, it is noted that the values reported by Vollenweider are considerably above the North American values reported by Uttormark et al. (1974). Typical values of watershed nutrient export coefficients are presented in Table 16. It is noted that while wetlands can act as sinks or sources of nutrients, depending on the season of the year, in general the net nutrient contribution from wetlands is considered to be zero (Sonzogni and Lee, 1974; Uttormark et al., 1974, Lee et al., 1975).

Table 16 indicates that several different nutrient export coefficients, varying widely in several cases, were available for each watershed land use category (i.e., 0.1 $g/m^2/yr$ (Sonzogni and Lee, 1974) vs. 0.03 $g/m^2/yr$ (US EPA, 1974c) for urban phosphorus export coefficient). As a result, the coefficients chosen to check the reported US OECD nutrient loadings are based largely on the experience of these reviewers and also on the regional nature of several of these values. For example, it was felt by these reviewers that the urban phosphorus and nitrogen export coefficients of Sonzogni and Lee (1974) represent a reasonable average of the values reported by Uttormark et al. (1974) and by the US EPA (1974c). The US EPA urban phosphorus and nitrogen export coefficients were based on studies done in 473 subdrainage areas in the eastern US. The coefficient of Sonzogni and Lee (1974) is also regional in that it was derived for the Lake Mendota, Wisconsin, watershed. However, it is more in agreement with that reported by Uttormark et al. (1974) than is the US EPA (1974c) value. While the coefficients of Uttormark et al. are also derived from studies confined mainly to the northeastern and upper midwestern US, they are also based on several studies done in the southern and western US and, therefore, represent more of a 'national average' than do the values of Sonzogni and Lee or the US EPA. Consequently, a certain bias was given to the values of Uttormark et al. (1974) as a reference national average value, even though they were based on studies confined largely to the upper midwestern and northeastern US.

A rural/agriculture phosphorus value of 0.05 $g/m^2/yr$ was taken as an average of the values of Sonzogni and Lee (1974) and both Uttormark et al. (1974) and the US EPA (1974c). A rural/agriculture nitrogen export coefficient of 0.05 $g/m^2/yr$ was used because of the agreement between the value of Sonzogni and Lee and that of Uttormark et al. The forest phosphorus export coefficient of Uttormark et al. was thought to be too high, based on the experience of these reviewers and on the "mostly forest" value reported by the US EPA. Consequently, the US EPA (1974c) forest phosphorus export coefficient of 0.01 $g/m^2/yr$ was used by these reviewers. A forest nitrogen export coefficient of

Table 16. TYPICAL VALUES OF WATERSHED NUTRIENT EXPORT COEFFICIENTS

Watershed Land Usage	Source: Sonzogni and Lee (1974)	Source: Uttormark *et al*. (1974)[a]	Source: US EPA (1974c)
	A. Total Phosphorus (g $P/m^2/yr$)		
Urban	0.1	0.15	0.03
Rural/Agriculture	0.07	0.03	0.03 (0.02)[b]
Forest	-	0.02	0.01 (0.02)[c]
Wetlands	Net nutrient contribution is considered to be zero.		
Other:			
Rainfall onto water body surface	0.02	-	-
Dry fallout onto water body surface	0.08	-	-
			mixed = 0.02[d]
	B. Total Nitrogen (g $N/m^2/yr$)		
Urban	0.5	0.5	0.8
Rural/Agriculture	0.5	0.5	1.0 (0.6)[b]
Forest	-	0.25	0.4 (0.4)[c]
Wetlands	Net nutrient contribution is considered to be zero.		
Other:			
Rainfall onto water body surface	0.8	-	-
Dry fallout onto water body surface	1.6	-	-
			mixed = 0.6[d]

[a]"Average" value indicated by Uttormark *et al*. (1974).

[b]Mostly agriculture; other types present.

[c]Mostly forest; other types present.

[d]Does not fit into any of the other watershed land use categories.

0.3 $g/m^2/yr$ was taken as an average of the values reported by Uttormark et al. and the US EPA. The one exception to these values is that the "low" nitrogen export coefficients reported by Uttormark et al. (1974) were used as a check on the reported nitrogen loadings of the US OECD water bodies located in the western US. These low values were used because most water bodies in the western US tend to be nitrogen-limited with respect to aquatic plant nutrient requirements. It was felt by these reviewers that the low nitrogen values were more accurate than the "average" values reported by Uttormark et al. (1974). These low nitrogen values were used for calculating the nitrogen loadings for Lakes Tahoe, Waldo, Sammamish and Washington.

The values for the nutrient contributions to the US OECD water bodies from precipitation and dry fallout directly onto the water body surface, if not indicated by the investigator, were taken from Sonzogni and Lee (1974). While precipitation and dry fallout nutrient contributions likely vary from location to location, the portion of nutrients contributed by precipitation or dry fallout onto a water body's surface was usually small, compared to the magnitude of the other input sources. Consequently, it was not considered a serious source of error to use the values reported by Sonzogni and Lee (1974).

A summary of the watershed land use nutrient export coefficients used by these reviewers as a check on the reported US OECD water body nutrient loadings is presented in Table 17.

Table 17. WATERSHED NUTRIENT EXPORT COEFFICIENTS USED TO CHECK US OECD NUTRIENT LOADINGS

Watershed Land Use	Watershed Export Coefficient ($g/m^2/yr$)
	A. Total Phosphorus
Urban	0.1
Rural/Agriculture	0.05
Forest	0.01
Other:	
Rainfall	0.02
Dry Falout	0.08
	B. Total Nitrogen
Urban	0.5 (0.25)[a]
Rural/Agriculture	0.5 (0.2)[a]
Forest	0.3 (0.1)[a]
Other:	
Rainfall	0.8
Dry Fallout	1.6

[a]Export coefficients used in calculating nitrogen loadings for US OECD water bodies in western US (i.e., Lakes Tahoe, Waldo, Sammamish and Washington).

In order to use these watershed land use and atmospheric nutrient export coefficients, the percentage of each of the four land use types in the watershed was determined from the data provided by the US OECD investigators. In some cases, an interpretation of a given watershed land usage type was used for this report if the US OECD investigator's description did not fit into any of the four watershed land use categories reported by Uttormark et al. (1974) (i.e., "residential," "commercial," "industrial," "public, semipublic transportation" and "mining" all being placed in the 'urban' category; "outdoor recreation" put into the 'forest' category, etc.). In general, the effect of the occasional liberal usage of watershed land use categories by these reviewers have tended to overestimate the nutrient loadings to the US OECD water bodies to some extent. In most cases, the investigator's reported watershed land usages conformed to the general categories defined by Uttormark et al. (1974). However, the methods employed in determining the watershed land usage patterns, or the sources of the watershed land usage data, if it was not directly determined, were usually not indicated by the US OECD investigators. Any other nutrient contribution values used in this portion of the report were those supplied by the US OECD investigators for their particular water bodies. These included wastewater discharges, groundwater inputs, spring inputs, nitrogen fixation (for nitrogen loading estimates) and marsh drainage.

The total phosphorus and total nitrogen loadings, as calculated using watershed land use nutrient runoff coefficients, are presented in Table 18. The US OECD investigator-indicated total phosphorus and total nitrogen loadings are included in Table 18 for comparison with the loadings derived from watershed land use nutrient export coefficients. The ratio of the export coefficient-derived nutrient loadings to the investigator-indicated loadings is also presented in Table 18. A ratio of one indicates agreement between the investigator-indicated nutrient loadings and the nutrient loadings calculated from watershed nutrient export coefficients. A ratio greater than one indicates the investigator-indicated nutrient loadings may have been underestimated, relative to the nutrient loading estimates obtained from the watershed land usage calculations. That is, the investigator-indicated nutrient loading is lower than the loading based on the watershed nutrient export coefficients listed in Table 17. Conversely, for a ratio less than one, the possibility of a nutrient loading overestimation is indicated.

According to Piwoni and Lee (1975) the nutrient loadings for Lakes Blackhawk, Camelot-Sherwood, Cox Hollow, Dutch Hollow, Redstone, Stewart, Twin Valley and Virginia were calculated using nutrient export coefficients derived by Sonzogni and Lee (1974). Since the nutrient export coefficients derived by Sonzogni and Lee (1974) are different for some land use types than those used by these reviewers, comparing the reported nutrient loadings for

Table 18. US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr) Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
A. PHOSPHORUS LOADINGS:								
Brownie (2)[d]	Watershed land usage data not available							
Calhoun (3)	Watershed land usage data not available							
Canadarago (5)[d]	2.8×10^6	6.02×10^5	4.3×10^6	6.02×10^5	7.6×10^5	1.2	0.8	1.5
Cayuga (6)	6.39×10^7	7.32×10^6 (Includes commercial, industrial mining, public and transportation)	5.95×10^7 (Includes active & inactive agriculture)	0	1.7×10^7	0.9	0.8	1.1
Cedar (7)	Watershed land usage data not available							
Dogfish (10)	0	0	0	5.9×10^3	2.9×10^4	0.1	0.02	5
George (12)	Watershed land usage data not available							
Harriet (13)	Watershed land usage data not available							

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr) Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
Isles (14)	Watershed land usage data not available							
Kerr Reservoir (Whole reservoir) (15)	2.34×10^7	2×10^8	3×10^8	1.2×10^8	3.2×10^7	4.0	4.0	1.0
Lamb (19)	0	0	0	1.6×10^4	4×10^4	0.14	0.03	4.7
Meander (21)	0	0	0	1.34×10^4	3.6×10^4	0.14	0.03	4.7
Mendota (22)	0	7.81×10^6	2.7×10^7	6.51×10^4	1.09×10^7 (Includes groundwater, baseflow, & storm drainage)	1.2	1.2	1.0
Michigan	Watershed land usage data not available							
Lower Lake Minnetonka	Watershed land usage data not available							
Potomac Estuary (entire estuary) (27)	4×10^9 (Median flow regime)	1.86×10^8	7.46×10^8	2.05×10^8	9.67×10^7	5.4	5.0	1.1

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr) Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
Sallie (32)	7.06×10^{6} -2.01×10^{7}	4.5×10^{6}	3.38×10^{7}	3.45×10^{6}	1.15×10^{6}	9.4-12.1	1.5-4.2	2.2-8.1
Sammamish (following diversion of sewage) (33)	5×10^{5}	2.75×10^{6}	3.75×10^{5}	2.15×10^{6}	2.6×10^{6}	0.4	0.7	0.6
Shagawa (34)	5.18×10^{6}	1.7×10^{6}	1.31×10^{5}	2×10^{6}	8.86×10^{5}	1.1	0.7	1.6
Tahoe (36)	0	2.88×10^{7}	0	4.72×10^{6}	5×10^{7}	0.17	0.05	3.4
Twin Lakes (East Twin & West Twin combined)								
1972 (39 & 43)	0	1.0×10^{5}	0	8.02×10^{3}	2.4×10^{5}	0.57	0.51 (0.53)[g]	1.1 (1.1)[g]
1973 (40 & 44)	0	1.0×10^{5}	0	8.02×10^{3}	2.4×10^{5}	0.57	0.40 (0.31)	1.4 (1.8)
1974 (41 & 45)	0	1.0×10^{5}	0	8.02×10^{3}	2.4×10^{5}	0.57	0.45 (0.54)	1.3 (1.1)

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr): Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
Waldo (48)	0	0	0	$5.2x10^5$	$2.88x10^6$	0.12	0.02	6.3
Washington (assumed 90 percent forest and 10 percent urban)								
1957 (49)	$5.7x10^7$	$1.61x10^7$	0	$1.45x10^7$	$8.8x10^6$	1.09	1.2	0.9
1964 (50)	$1.04x10^8$	$1.61x10^7$	0	$1.45x10^7$	$8.8x10^6$	1.63	2.3	0.7
1971 (51)	0	$1.61x10^7$	0	$1.45x10^7$	$8.8x10^6$	0.45	0.43	1.0
1974 (52)	0	$1.61x10^7$	0	$1.45x10^7$	$8.8x10^6$	0.45	0.47	1.0
Weir (53)	0	$3.68x10^5$	$1.7x10^6$	$8.74x10^4$	$3.5x10^6$ (includes septic tanks)	0.24	0.14	1.7
Wingra (54)	0	$1.05x10^6$	0	$3.13x10^4$	$2.19x10^5$ (includes spring flow)	0.93	0.9	1.0

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr) Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
B. NITROGEN LOADINGS:[e]								
Brownie (2)	Nitrogen loadings not determined							
Calhoun (3)	Nitrogen loadings not determined							
Canadarago (5)	7.8×10^6	3.01×10^6	4.3×10^7	1.81×10^7	1.79×10^7	11.8	18.0	0.7
Cayuga	1.68×10^8	3.66×10^7	5.95×10^8	1.76×10^8	4.01×10^8	8.1	14.3 (does not include organic nitrogen)	0.6
Cedar (7)	Nitrogen loadings not determined							
Dogfish	Nitrogen loadings not determined							
George (12)	Watershed land usage data not available							
Harriet (13)	Nitrogen loadings not determined							

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr)				Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
		Urban	Rural	Forest	Other[c]			
Isles (14)	Watershed land usage data not available							
Kerr Reservoir (Whole reservoir) (15)	$2.34x10^7$	$2x10^8$	$3x10^8$	$1.2x10^8$	$3.2x10^7$	4.0	4.0	1.0
Lamb (19)	0	0	0	$1.6x10^4$	$4x10^4$	0.14	0.03	4.7
Meander (21)	0	0	0	$1.34x10^4$	$3.6x10^4$	0.14	0.03	4.7
Mendota (22)	0	$7.81x10^6$	$2.7x10^7$	$6.51x10^4$	$1.09x10^7$ (Includes groundwater, baseflow, & storm drainage)	1.2	1.2	1.0
Michigan	Watershed land usage data not available							
Lower Lake Minnetonka	Watershed land usage data not available							
Potomac Estuary (entire estuary) (27)	4×10^9 (Median flow regime)	$1.86x10^8$	$7.46x10^8$	$2.05x10^8$	$9.67x10^7$	5.4	5.0	1.1

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr): Urban	Rural	Forest	Other[c]	Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
Sallie (32)	$5.59x10^{6}$ - $1.14x10^{7}$	$2.25x10^{7}$	$3.38x10^{8}$	$1.04x10^{8}$	$1.59x10^{7}$ - $2.13x10^{7}$	91.6-93.8	2.8-3.0	30-34
Sammamish[f] (following sewage diversion) (33)	Unknown	$6.88x10^{6}$	$1.5x10^{6}$	$2.15x10^{7}$	$4.72x10^{7}$	-	13.0	-
Shagawa (34)	$1.93x10^{7}$	$8.48x10^{6}$	$1.3x10^{6}$	$6.03x10^{7}$	$1.84x10^{7}$ - $2.37x10^{7}$	11.7-12.3	7.8	1.5-1.6
Tahoe[f] (36)	0	$7.2x10^{7}$	0	$4.72x10^{7}$	$9.04x10^{8}$	2.0	0.52	3.8
Twin Lakes[f] (East Twin Lake & West Twin Lake Combined)								
1972 (39 & 43)	0	$5.01x10^{5}$	0	$8.02x10^{4}$	$5.27x10^{6}$	9.6	22.6	0.4
1973 (40 & 44)	0	$5.01x10^{5}$	0	$8.02x10^{4}$	$5.02x10^{6}$	9.2	16.8 (does not include organic nitrogen)	0.5

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr)				Loadings Calculated via Export Coefficients (g/m^2yr)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
		Urban	Rural	Forest	Other[c]			
Waldo[f] (48)	0	0	0	$5.2x10^6$	$5.4x10^7$	2.2	0.33	6.6
Washington (assumed 10 percent urban and 90 percent forest)								
1957 (49)	$2.01x10^8$	$4.02x10^7$	0	$1.45x10^8$	$2.07x10^8$	6.7	19.2	0.3
1964 (50)	$2.71x10^8$	$4.02x10^7$	0	$1.45x10^8$	$2.07x10^8$	7.5	7.8	1.0
1971 (51)	0	$4.02x10^7$	0	$1.45x10^8$	$2.07x10^8$	4.4	4.6	1.0
1974 (52)	0	$4.02x10^7$	0	$1.45x10^8$	$2.07x10^8$	4.4	4.4	1.0
Weir (53)	0	$1.84x10^6$	$1.68x10^7$ (includes pasture)	$2.64x10^6$	$5.3x10^7$ (includes septic tanks)	3.1	2.6	1.2

Table 18 (continued). US OECD NUTRIENT LOADINGS CALCULATED USING WATERSHED NUTRIENT EXPORT COEFFICIENTS

Water Body	Point-Source Loading[a] (g/yr)	Non-Point Source Loading[b] (g/yr)				Loadings Calculated via Export Coefficients ($g/m^2 yr$)	Investigator-Indicated Loadings ($g/m^2/yr$)[a]	Ratio of Export Coefficient Loadings to Investigator-Indicated Loadings
		Urban	Rural	Forest	Other[c]			
Wingra (54)	0	5.24×10^6	0	9.38×10^5	7.57×10^6 (includes spring flow)	9.8	5.1	1.9

[a]Based on investigator's estimates.

[b]Watershed land usage as defined by Uttormark _et al_. (1974) and indicated by the investigator.

[c]As indicated by the investigator. Precipitation and dry fallout nutrient inputs, if not indicated by the investigator, were calculated using the nutrient coefficients given in Sonzogni and Lee (1974). Other loadings are as indicated in the table.

[d]Identification number for Figures 15 and 16 (see Table 14).

[e]Nitrogen loadings are comprised of inorganic nitrogen (i.e., $NH_4^+ + NO_3^- + NO_2^-$ as N) plus organic nitrogen, unless otherwise indicated.

[f]The "low" nitrogen export coefficient of Uttormark _et al_. (1974) used to determine the nitrogen loading estimate.

[g]Data in parentheses represent data received by these investigators from the principal investigator subsequent to the completion of this report. Figures 15 and 16 are based on the original data reported by the investigator and do not reflect the changes indicated above. Examination of this subsequent data indicates the phosphorus loads were originally underestimated; however, there were no significant changes in the overall conclusions concerning the Twin Lakes as a result of these altered loads.

these water bodies with those calculated using the nutrient export coefficients in Table 17 would obviously indicate an error in the reported nutrient loadings. Further, it was also felt by these reviewers that land use export coefficients calculated for a specific watershed are likely more accurate than the average values used in these calculations. Consequently, these water bodies were not included in Table 18 as it would be incorrect to check their nutrient loadings in this manner. Lake Waldo's reported nutrient loadings are based on an average of several indirect methods, including land use export coefficients derived for the Upper Klamath (Powers *et al.*, 1975). However, since more than one method was used by Powers *et al.* to calculate Lake Waldo's nutrient loading and because the value obtained using the export coefficient was similar to the value obtained with the other indirect methods, this water body was retained in Table 18.

The watershed land use-derived loading estimates for phosphorus and nitrogen are compared with the US OECD investigator-indicated loadings in Figures 15 and 16, respectively. The various lines and the shaded zone in Figures 15 and 16 have the same meaning as those in Figure 14. Figures 15 and 16 will be discussed in connection with the Vollenweider loadings diagrams presented in following sections of this report.

Comparison of Phosphorus Loadings Derived From Vollenwider Relationship With Loadings Derived From Watershed Phosphorus Export Coefficients.

The phosphorus loadings predicted by Vollenweider's relationship in Equation 26 may be compared with the loadings predicted with the use of watershed land use phosphorus export coefficients. If they are similar, one can have some degree of confidence that their use for determining the correct value for the phosphorus loadings was somewhat justified. If they disagree to any major extent, then one would have to question the use of one or both of these approaches for predicting the 'correct' phosphorus loadings to the US OECD water bodies. Such a comparison was made with the US OECD eutrophication study data. The predicted phosphorus loadings, using the Vollenweider relationship expressed in Equation 26 and the watershed land use phosphorus export coefficients, as well as the ratio of the former to the latter, is presented in Table 19. The results are presented graphically in Figure 17. The various lines and the shaded zone in Figure 17 have the same meaning as in Figure 14. If a data range was reported for a water body, the mean value was used in all calculations.

Examination of Figure 17 shows reasonably good agreement between the phosphorus loadings predicted for the US OECD water bodies using the Vollenweider relationship (Equation 26) and those predicted using watershed phosphorus export coefficients. Most of the phosphorus loadings predicted using Equation 26 are

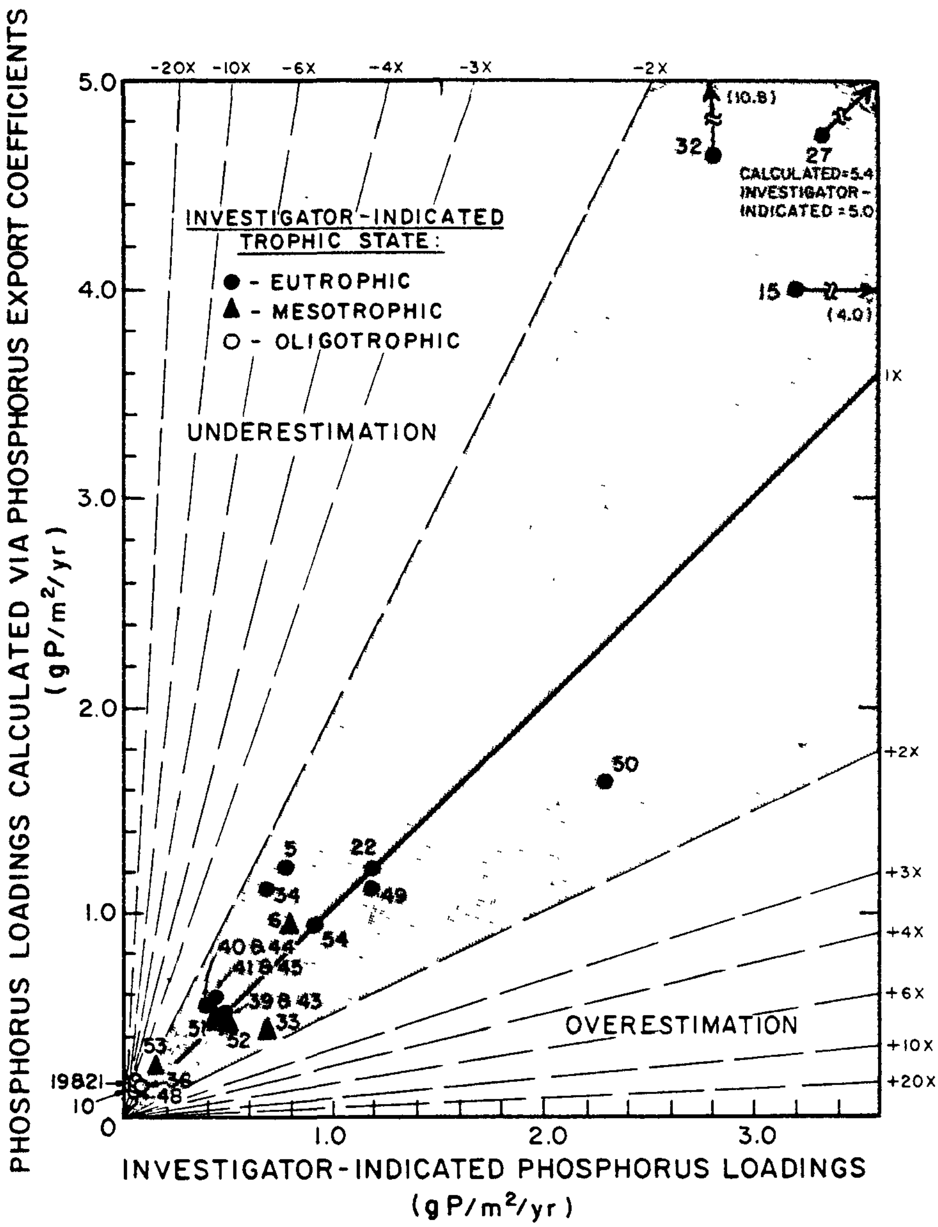


Figure 15. Evaluation of Estimates of US OECD Water Body Nutrient Loadings: Watershed Land Use Phosphorus Coefficient Calculations

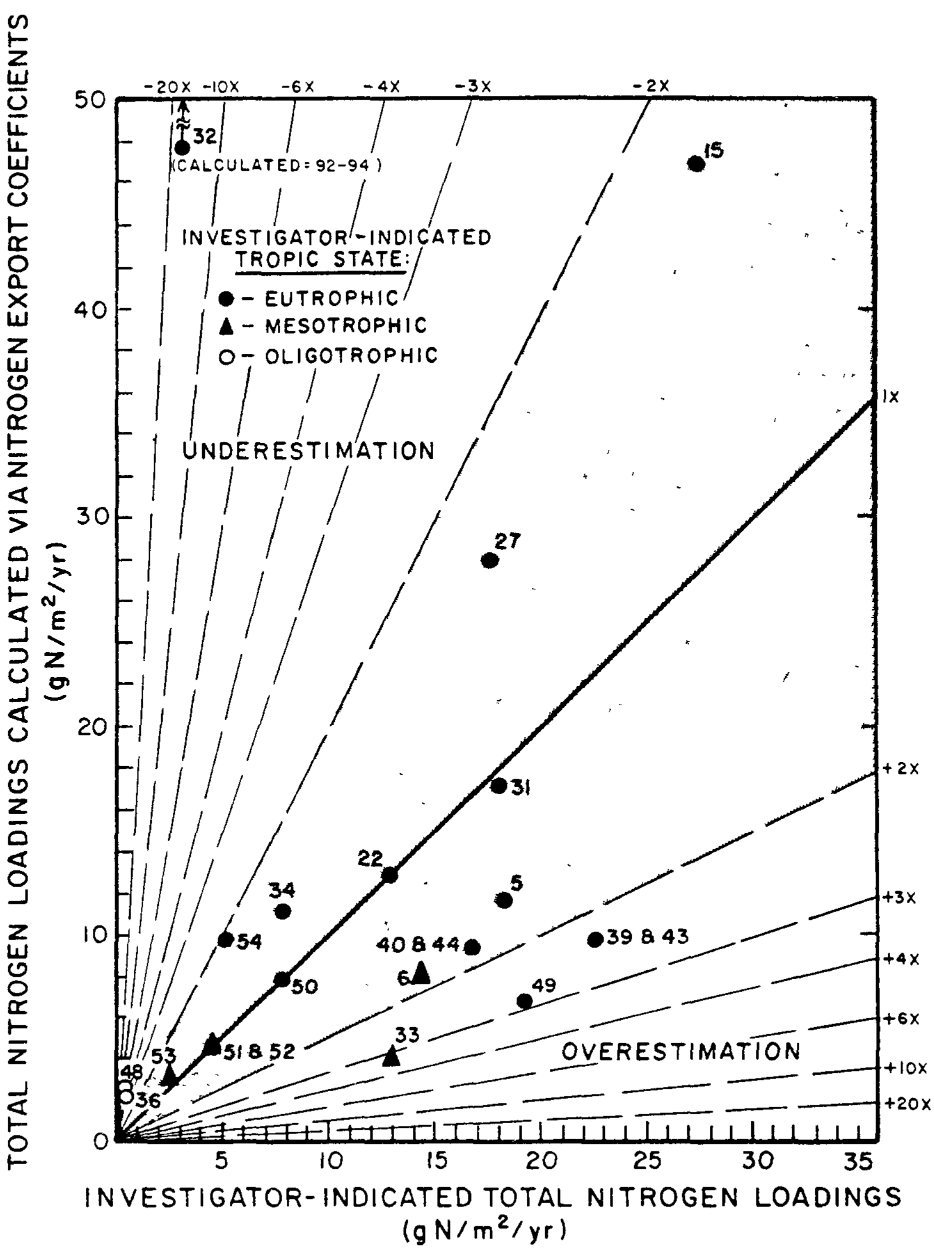


Figure 16. Evaluation of Estimates of US OECD Water Body Nutrient Loadings: Watershed Land Use Nitrogen Export Coefficient Calculations

Table 19. COMPARISON OF PHOSPHORUS LOADINGS DERIVED FROM WATERSHED EXPORT COEFFICIENTS WITH LOADINGS PREDICTED BY VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loadings Predicted with Vollenweider's Relationship (Equation 25)[b] ($g/m^2/yr$)	Phosphorus Loadings Predicted with Watershed Phosphorus Export Coefficients[c] ($g/m^2/yr$)	Ratio of Vollenweider-Derived to Export Coefficient-Derived Loadings
Calhoun (3)[d]	E	0.9[e]	-	-
Canadarago (5)	E	0.9-1.1	1.2	0.8-0.9
Cayuga (6)	M	0.50	0.9	0.6
Cedar (7)	E	0.3[e]	-	-
Dogfish (10)	O	0.02[e]	0.1	0.2
George (12)	O-M	0.07	-	-
Harriet (13)	E	0.6[e]	-	-
Isles (14)	E	0.9[e]	-	-
Kerr Reservoir[f] (15)	E-M	-	-	-
Lamb (19)	O	0.05-0.06	0.14	0.4
Meander (21)	O	0.04-0.06	0.14	0.3-0.4
Mendota (22)	E	1.2	1.2	1.0
Michigan Open Waters (23A & B)	O	0.1-0.2	-	-
Lower Lake Minnetonka[f]	E→M	-	-	-
Potomac Estuary[f] (27)	U-E	-	5.4	-
Sallie (32)	E	2.6-4.7	9.4-12.1	0.2-0.5

Table 19 (continued). COMPARISON OF PHOSPHORUS LOADINGS DERIVED FROM WATERSHED EXPORT COEFFICIENTS WITH LOADINGS PREDICTED BY VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loadings Predicted with Vollenweider's Relationship (Equation 25)[b] ($g/m^2/yr$)	Phosphorus Loadings Predicted with Watershed Phosphorus Export Coefficients[c] ($g/m^2/yr$)	Ratio of Vollenweider-Derived to Export Coefficient-Derived Loadings
Sammamish (33)	M	0.4	0.7	0.6
Shagawa (34)	E	0.8	1.1	0.7
Tahoe (36)	U-O	0.03	0.2	0.2
Twin Lakes (East Twin & West Twin)				
1972 (39 & 43)	E	0.7-0.9 (0.6)[h]	0.6(0.6)[h]	1.2-1.5(1.0)[h]
1973 (40 & 44)	E	0.6-0.9 (0.5)	0.6(0.6)	1.0-1.5(0.8)
1974 (41 & 45)	E	0.9-1.4 (0.6)	0.6(0.6)	1.5-2.3(1.0)
Waldo (48)	U-O	<0.05[g]	0.12	<0.4
Washington				
1957 (49)	E	0.8	1.1	0.7
1964 (50)	E	2.3	1.6	1.4
1971 (51)	M	0.6	0.45	1.3
Weir (53)	M	0.12	0.24	0.5
Wingra (54)	E	0.7	0.9	0.8

EXPLANATION:

[a]Investigator-indicated trophic state:
E = eutrophic   O = oligotrophic
M = mesotrophic   U = ultra

Table 19 (continued). COMPARISON OF PHOSPHORUS LOADINGS DERIVED FROM WATERSHED EXPORT COEFFICIENTS WITH LOADINGS PREDICTED BY VOLLENWEIDER'S MEAN PHOSPHORUS/INFLUENT PHOSPHORUS AND HYDRAULIC RESIDENCE TIME RELATIONSHIP

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[b]Phosphorus loadings calculated using the investigator-indicated mean phosphorus concentrations and hydraulic loading ($\bar{z}/\tau_{\omega}$) data, as applied in Equation 25.

[c]Phosphorus loadings calculated using the watershed nutrient export coefficients cited in Table 17. Point sources and any other additional nutrient input sources used in the calculations were those supplied by the US OECD investigators for their respective water bodies.

[d]Identification number for Figure 17 (see Table 14).

[e]The mean phosphorus concentrations used in Equation 25 were the average summer surface values.

[f]Mean phosphorus concentrations were reported for the arms or sub-basins of these water bodies, while the watershed land usage patterns were reported for the entire watershed. Because of mixing of nutrients added to the water body as a whole, as well as morphological and hydrological differences between the sub-basins, it is not possible to calculate phosphorus loadings based on watershed land use nutrient export coefficients for these water bodies.

[g]The mean phosphorus concentrations used in Equation 25 were derived from annual August average values.

[h]Data in parentheses represent calculations based on data received by these reviewers from the principal investigator subsequent to the completion of this report. Figure 17 is based on the original data reported by the investigator and does not reflect the changes indicated above. However, examination of this subsequent data indicated there were no significant changes in the overall conclusions concerning the Twin Lakes as a result of these altered values.

Dash (-) indicates data not available.

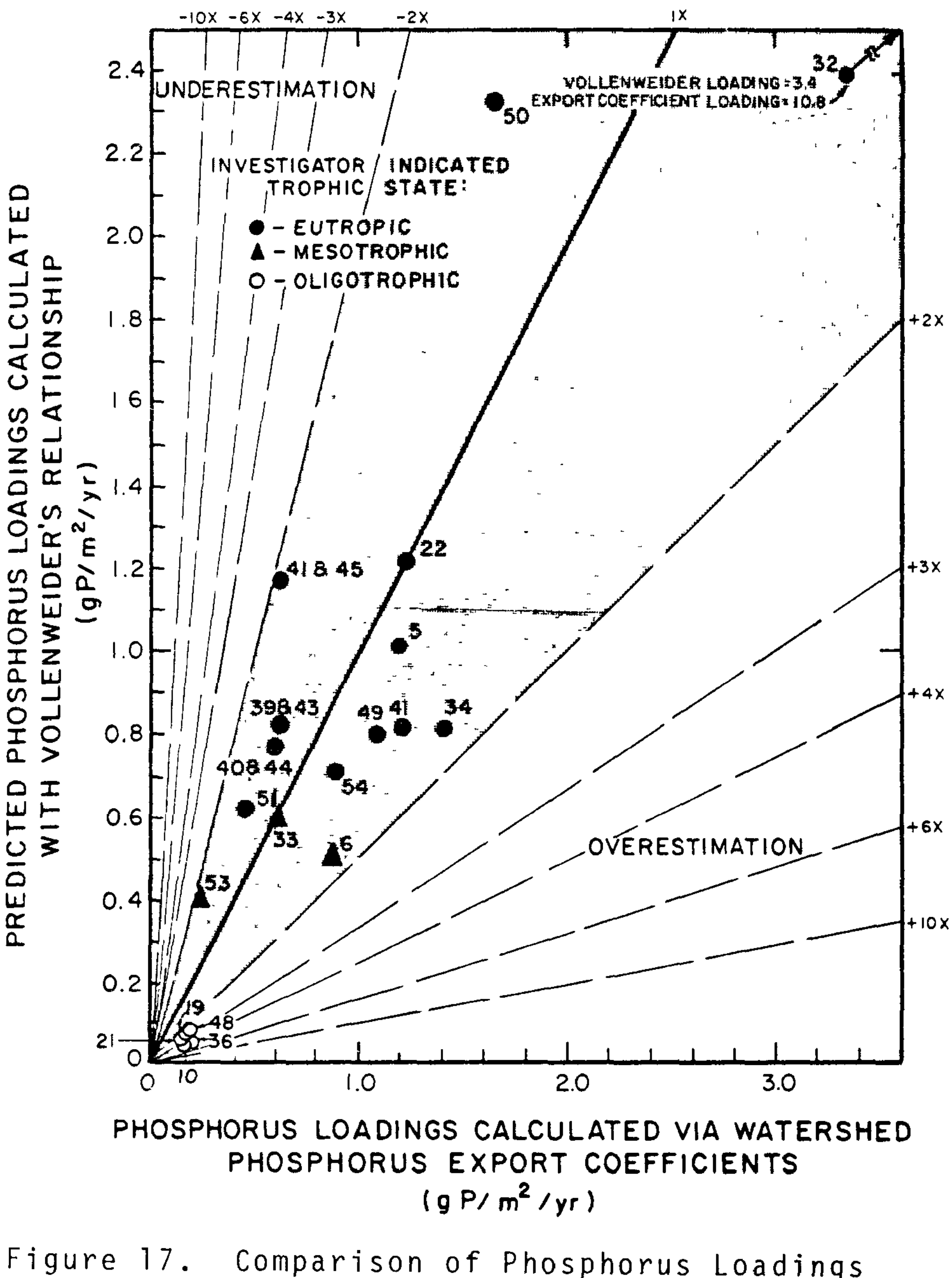

Figure 17. Comparison of Phosphorus Loadings Derived from Watershed Export Coefficients with Loadings Derived from Vollenweider Mean Phosphorus/Influent Phosphorus and Hydraulic Residence Time Relationship

within two-fold of the loadings predicted with nutrient export coefficients Given the different components considered in these two approaches, a phosphorus loading discrepancy of two-fold or less between these two methods was considered by these reviewers to be a reasonably good agreement for the water bodies for which adequate data were available. The results of Figure 17 and Table 19 will also be discussed in connection with the Vollenweider loading diagrams presented in subsequent sections of this report.

SECTION VII

US OECD EUTROPHICATION STUDY PHOSPHORUS DATA:

AS APPLIED IN INITIAL VOLLENWEIDER PHOSPHORUS LOADING AND MEAN DEPTH/HYDRAULIC RESIDENCE TIME RELATIONSHIP

With the possible phosphorus loading discrepancies indicated in the relationships discussed in the previous section (i.e., Figures 14-16), it is now appropriate to return to the major focus of the US OECD eutrophication study and examine the phosphorus loading-trophic response relationships in the US OECD water bodies, as expressed by the Vollenweider phosphorus loading criteria and other models.

The Vollenweider diagram of total phosphorus loading and the ratio of mean depth to hydraulic residence time, as originally developed (Vollenweider, 1975a), containing the US OECD water bodies for the years for which data were available is presented in Figure 18. This is the phosphorus loading diagram which serves as the basis of the US EPA's Quality Criteria for Water (US EPA, 1976a) for determining critical phosphorus loads for US lakes and impoundments. The pertinent US OECD data are presented in Table 20. If a data range was reported for a water body, the mean value was used in all calculations. Data were not available for all water bodies for all time periods. An example is Dogfish Lake. Nutrient data were available only for 1972. Consequently, in Figure 18, only Dogfish Lake - 1972 (Identification Number 10) is presented. Refer to Table 14 for identification of any water bodies and/or time periods not included in a given table or figure in this report.

Examination of Figure 18 shows good agreement between the trophic states of the US OECD eutrophication study water bodies, as indicated by their position on the Vollenweider phosphorus loading diagram (based on their reported phosphorus loadings and mean depth/hydraulic residence time characteristics), and the trophic states indicated by their principal investigators. Only a few water bodies show anomalies between the predicted and reported trophic states. These anomalies will be discussed shortly. The small number of US OECD water bodies showing disagreement between the Vollenweider phosphorus loading diagram-indicated trophic state associations and the investigator-indicated trophic states support the validity of the Vollenweider phosphorus load-

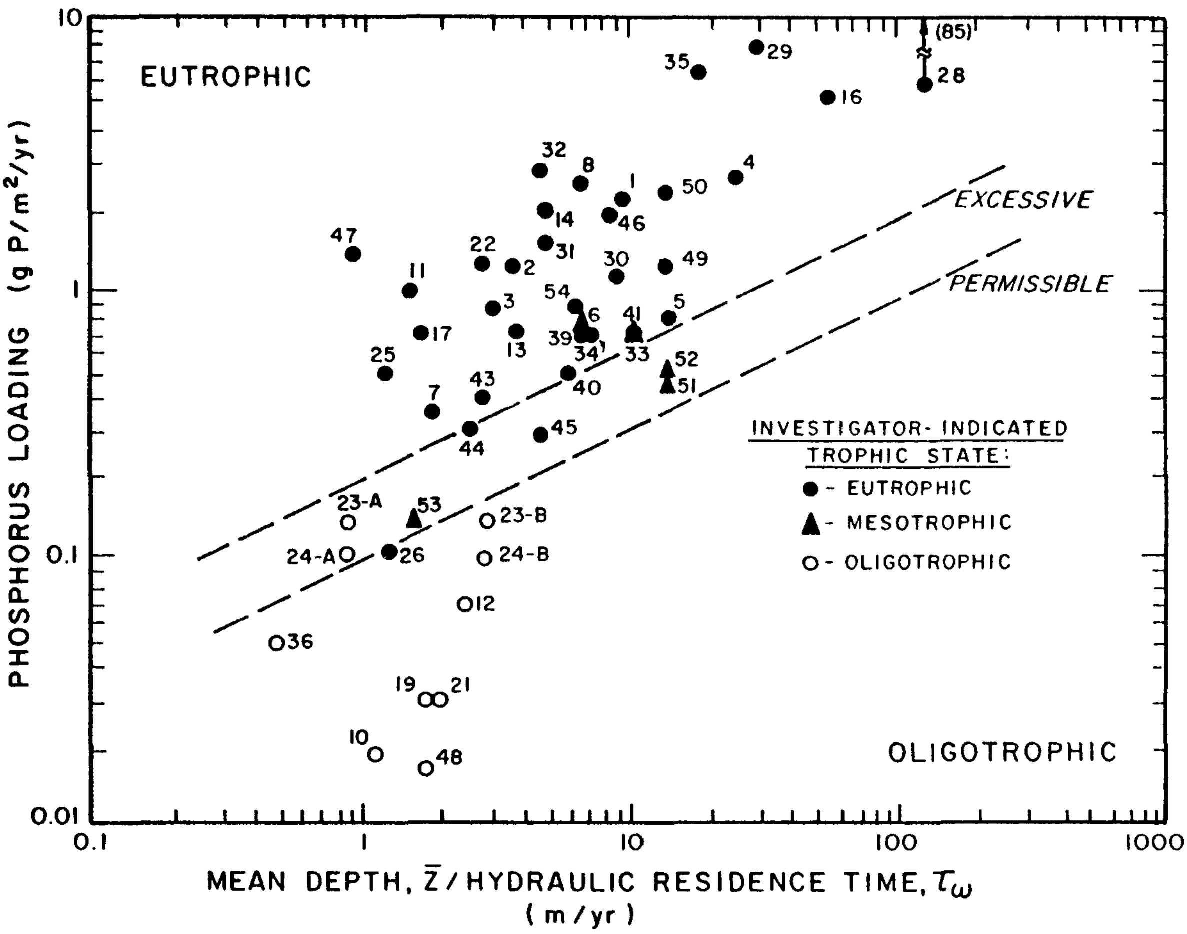

Figure 18. US OECD Data Applied to Initial Vollenweider Phosphorus Loading and Mean Depth/Hydraulic Residence Time Relationship

Table 20. PHOSPHORUS AND NITROGEN LOADINGS, MEAN DEPTHS ($\bar{z}$) AND HYDRAULIC RESIDENCE TIMES (τ_{ω}) FOR US OECD WATER BODIES

Water Body	Trophic State[a]	Mean Depth, $\bar{z}$ (m)[b]	Hydraulic Residence Time, τ_{ω} (yr)[c]	Total Phosphorus Loadings (g P/m^2/yr)[d]	Total Nitrogen Loadings (g N/m^2/yr)[e]
Blackhawk (1)[f]	E	4.9	0.5	2.1-2.3	23.4
Brownie (2)	E	6.8	2.0	1.18	-
Calhoun (3)	E	10.6	3.6	0.86	-
Camelot-Sherwood Complex (4)	E	3	0.09-0.14	2.4-2.7	34.6
Canadarago (5)	E	7.7	0.6	0.8	18
Cayuga (6)	M	54	8.6	0.8	14.3[g]
Cedar (7)	E	6.1	3.3	0.35	-
Cox Hollow (8)	E	3.8	0.5-0.7	1.6-2.1	19.1
Dogfish (10)	O	4.0	3.5	0.02	-
Dutch Hollow(11)	E	3	1.8	1.0	10.4
George (12)	O-M	18	8.0	0.07	1.8
Harriet (13)	E	8.8	2.4	0.71	-
Isles (14)	E	2.7	0.6	2.03	-
Kerr Reservoir	E-M				
Roanoke Arm(16)	-	10.3	0.2	5.2	36.2
Nutbush Arm(17)	-	8.2	5.1	0.7	2.4
Lamb (19)	O	4	2.3	0.03	-

Table 20 (continued). PHOSPHORUS AND NITROGEN LOADINGS, MEAN DEPTHS ($\bar{z}$) AND HYDRAULIC RESIDENCE TIMES (τ_{ω}) FOR US OECD WATER BODIES.

Water Body		Trophic State[a]	Mean Depth, $\bar{z}$ (m)[b]	Hydraulic Residence Time, τ_{ω} (yr)[c]	Total Phosphorus Loadings (g P/m^2/yr)[d]	Total Nitrogen Loadings (g N/m^2/yr)[e]
Meander	(21)	O	5.0	2.7	0.03	-
Mendota	(22)	E	12	4.5	1.2	13
Michigan (open waters)						
1971	(23 A & B)	O	84	30-100	0.14	-
1974	(24 A & B)	O	84	30-100	0.10	1.3
Lower Lake Minnetonka						
1969	(25)	E	8.3	6.3	0.5	-
1973	(26)	E→M	8.3	6.3	0.1(0.2)[h]	-
Potomac		U-E				
Upper	(28)	-	4.8	0.04	85	288
Middle	(29)	-	5.1	0.18	8	32
Lower	(30)	-	7.2	0.85	1.2	2.5
Redstone	(31)	E	4.3	0.7-1.0	1.4-1.7	18.1
Sallie	(32)	E	6.4	1.1-1.8	1.5-4.2	2.8-3.0
Sammamish	(33)	M	18	1.8	0.7	13
Shagawa	(34)	E	5.7	0.8	0.7	7.8

Table 20 (continued). PHOSPHORUS AND NITROGEN LOADINGS, MEAN DEPTHS ($\bar{z}$) AND HYDRAULIC RESIDENCE TIMES (τ_ω) FOR US OECD WATER BODIES.

Water Body		Trophic State[a]	Mean Depth, $\bar{z}$ (m)[b]	Hydraulic Residence Time, τ_ω (yr)[c]	Total Phosphorus Loadings (g P/m^2/yr)[d]	Total Nitrogen Loadings (g N/m^2/yr)[e]
Stewart	(35)	E	1.9	0.1	4.8-8.0	73.6
Tahoe	(36)	U-O	313	700	0.05	0.52
East Twin						
1972	(39)	E	5.0	0.8	0.7 (0.7)[h]	31.4[g]
1973	(40)	E	5.0	0.9	0.5 (0.5)	19.3[g]
1974	(41)	E	5.0	0.5	0.7 (0.8)	-
West Twin						
1972	(42)	E	4.34	1.6	0.4 (0.4)[h]	16[g]
1973	(43)	E	4.34	1.8	0.3 (0.2)	15[g]
1974	(44)	E	4.34	1.0	0.3 (0.3)	-
Twin Valley	(46)	E	3.8	0.4-0.5	1.7-2.0	17.4
Virginia	(47)	E	1.7	0.9-2.8	1.2-1.5	18.3
Waldo	(48)	U-O	36	21	0.017	0.33
Washington						
1957	(49)	E	33	2.4	1.2	19.2
1964	(50)	E	33	2.4	2.3	7.8
1971	(51)	M	33	2.4	0.43	4.6
1974	(52)	M	33	2.4	0.47	4.4

Table 20 (continued). PHOSPHORUS AND NITROGEN LOADINGS, MEAN DEPTHS ($\bar{z}$) AND HYDRAULIC RESIDENCE TIMES (τ_ω) FOR US OECD WATER BODIES.

Water Body		Trophic State[a]	Mean Depth, $\bar{z}$ (m)[b]	Hydraulic Residence Time, τ_ω (yr)[c]	Total Phosphorus Loadings (g P/m^2/yr)[d]	Total Nitrogen Loadings (g N/m^2/yr)[e]
Weir	(53)	M	6.3	4.2	0.14	2.6
Wingra	(54)	E	2.4	0.4	0.9	5.14

EXPLANATION:

[a]Investigator-indicated trophic state: E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

[b]Mean depth ($\bar{z}$) = water body volume (m^3)/water body surface area (m^2).

[c]Hydraulic residence time (τ_ω) = water body volume (m^3)/annual inflow volume (m^3/yr).

[d]Based on investigator's estimates.

[e]Based on investigator's estimates. Total nitrogen loading consists of inorganic nitrogen (i.e., $NH_4^+ + NO_3^- + NO_2^- - N$) + organic nitrogen, unless otherwise indicated.

[f]Identification number for Figures 18, 19 and 21 (See Table 14)

[g]Does not include organic nitrogen.

[h]Data in parentheses represents data received by these reviewers subsequent to the completion of this report. Figures 18 and 19 are based on the original data supplied by the investigators and do not reflect these revised values. Examination of the data indicated no significant changes in the overall conclusions concerning these water bodies.

Dash (-) indicates data not available.

ing relationship in establishing trophic state associations and critical phosphorus loading levels for US lakes and impoundments (i.e., a level which could produce problem algal blooms in water bodies).

For the purposes of this section of the report, agreement or lack of agreement with the Vollenweider relationship is based on whether the investigator-indicated trophic state is appropriate, compared with the trophic conditions that Vollenweider and other US OECD investigators have reported for other water bodies with similar phosphorus loadings and hydrologic and morphologic characteristics, (i.e., does a lake designated as eutrophic by the US OECD investigator hold a position on the Vollenweider loading curve similar to those held by other eutrophic lakes?). No attempt is being made at this time to further refine this relationship. If it is completely valid, then lakes with the greater displacement from the permissible phosphorus loading line should be more highly eutrophic. In general, this seems to be the case for many of the US OECD eutrophication study water bodies. This point will be discussed further in a subsequent section of this report.

AS APPLIED IN MODIFIED VOLLENWEIDER PHOSPHORUS LOADING AND MEAN DEPTH/HYDRAULIC RESIDENCE TIME RELATIONSHIP

Vollenweider's modified phosphorus loading and mean depth/hydraulic residence time diagram is presented with the US OECD eutrophication study data in Figure 19. As mentioned in an earlier section, this modified Vollenweider phosphorus loading diagram is identical to his original phosphorus loading and mean depth/hydraulic residence time diagram (Figure 18) except that the boundary conditions have been altered. According to Vollenweider (1975a), these modified boundary conditions are more indicative of the true phosphorus assimilative capacity of water bodies than were his original boundary conditions (Figure 18). These altered permissible and excessive loading lines (Figure 19) make a difference in the trophic zone designation of the loading diagram, lowering the permissible and excessive phosphorus loading limits for some range of $\bar{z}/\tau_{\omega}$ values and raising them for other values of $\bar{z}/\tau_{\omega}$. The original and modified Vollenweider phosphorus load and mean depth/hydraulic residence time loading diagrams are superimposed in Figure 20 to illustrate the differences in trophic zone designations.

Examination of Figure 20 shows the effect of the modified boundary conditions is to indicate a lower apparent phosphorus assimilative capacity (i.e., a lower permissible and excessive loading line) on the modified loading diagram (Figure 19) for water bodies with a $\bar{z}/\tau_{\omega}$ value of between approximately 2 to 50, relative to the original Vollenweider phosphorus loading diagram (Figure 18). Below a $\bar{z}/\tau_{\omega}$ value of 2, the phosphorus assimilative capacity becomes constant in the modified Vollenweider phosphorus loading diagram. The excessive and permissible loading boundary conditions increase in the modified Vollenweider phosphorus loading diagram above a $\bar{z}/\tau_{\omega}$ value of about 50. This

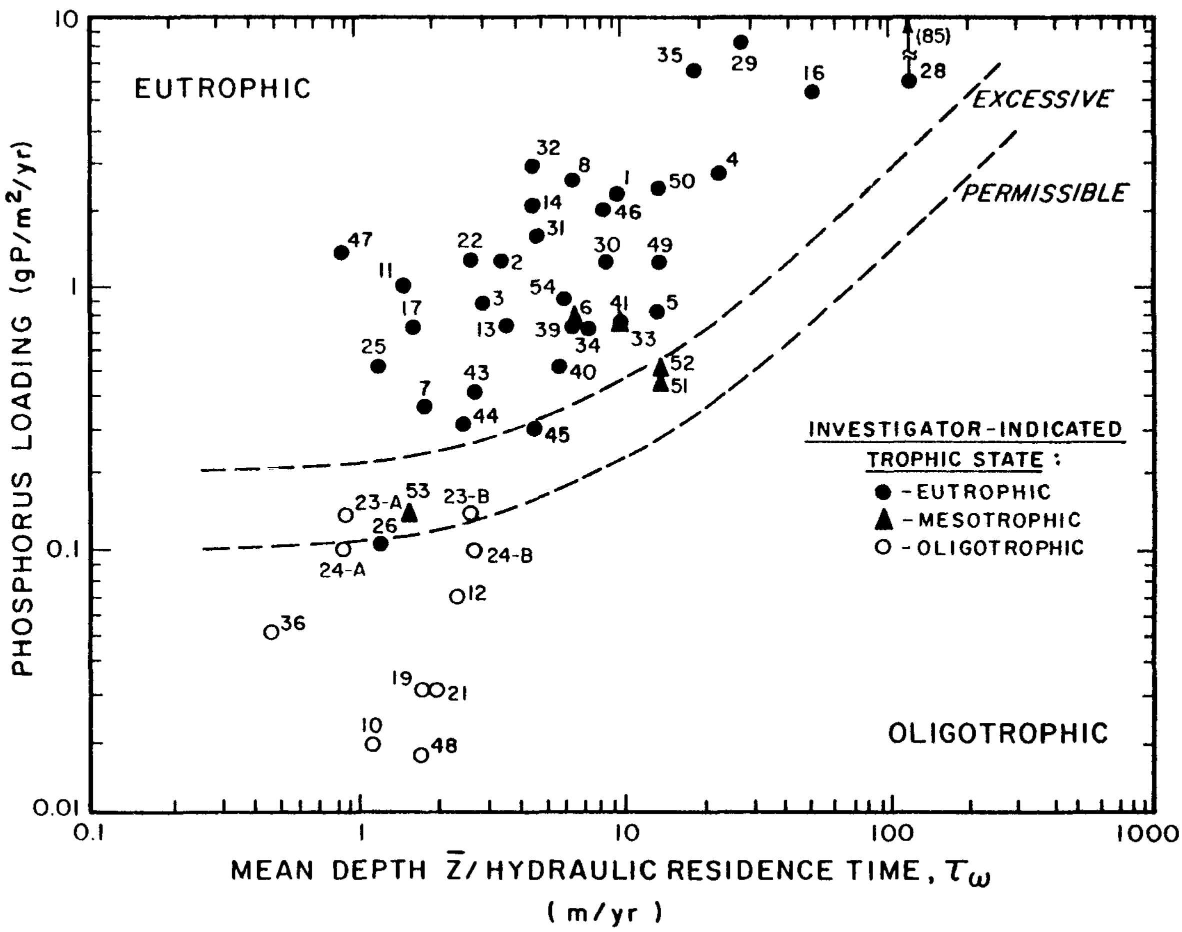

Figure 19. US OECD Data Applied to Modified Vollenweider Phosphorus Loading and Mean Depth/Hydraulic Residence Time Relationship

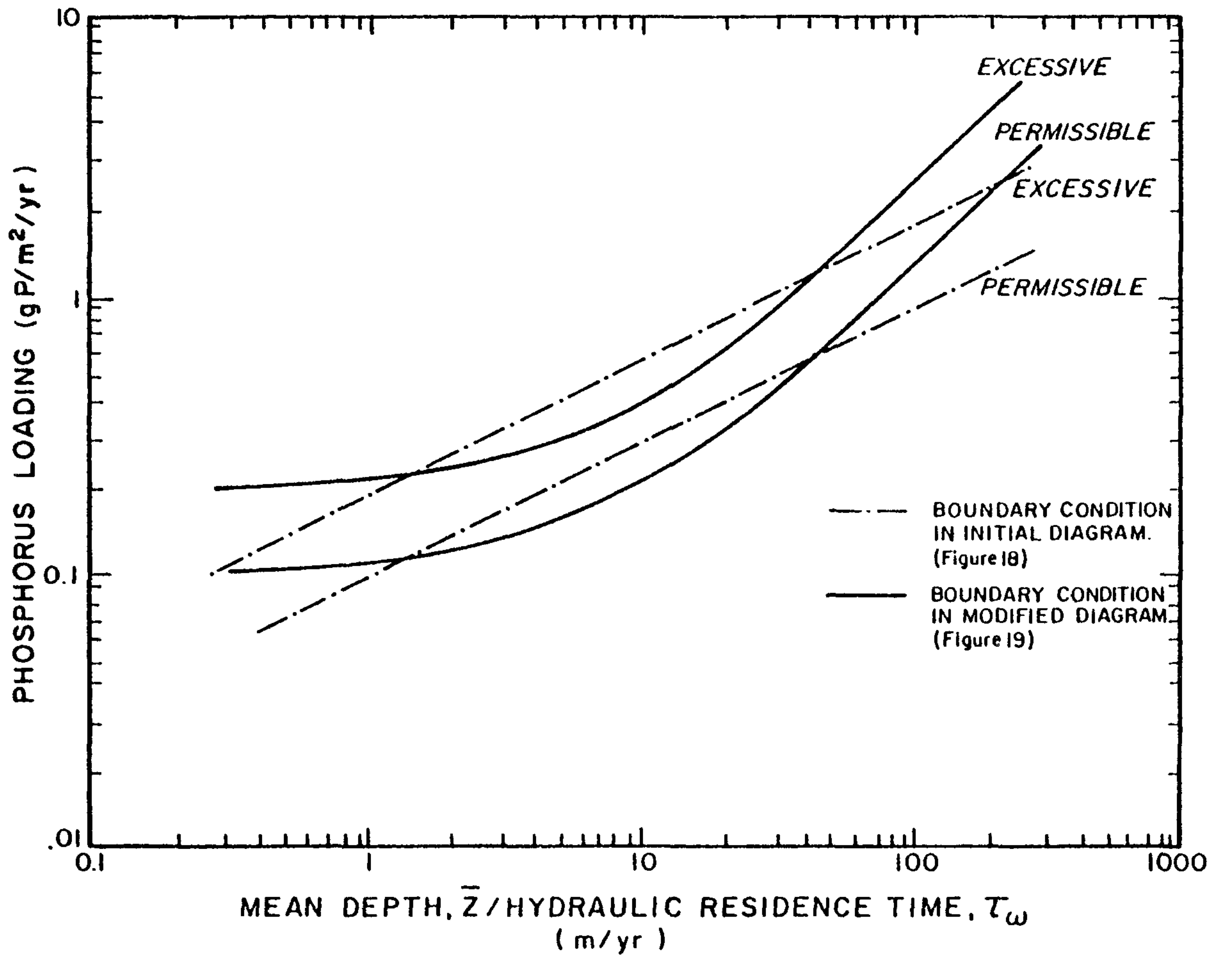

Figure 20. Comparison of Permissible and Excessive Loading Lines in Initial and Modified Vollenweider Phosphorus Loading Diagram.

increase in phosphorus loading tolerance illustrates the effects of either a great depth or a very rapid hydraulic flushing time on increasing the relative phosphorus assimilative capacity of a water body. A great depth in a water body usually indicates a large volume of water, with a likely high degree of dilution of input nutrients and reduced phosphorus return from the sediments, and gives the water body a high phosphorus assimilative capacity. Conversely, a very rapid flushing rate usually indicates that the nutrients are being washed out of the water body approximately as rapidly as they are being added to it, giving the water body a higher phosphorus assimilative capacity than water bodies with a lower $\bar{z}/\tau_\omega$ value.

Figure 19 represents one of the major thrusts of the US OECD eutrophication study. It demonstrates the relationship between the phosphorus loadings and trophic conditions of the US OECD water bodies, as modified by their hydraulic loading, q_s. This is based on their associations on the loading diagram with water bodies of similar z/τ_ω ($=q_s$) characteristics and phosphorus loads. It also establishes the permissible and excessive phosphorus loading levels for these water bodies. Figure 19 indicates that only Lakes Cayuga (6), Lower Minnetonka (26), and Sammamish (33) have predicted trophic states which are in disagreement with the trophic state reported by the respective US OECD investigator (Appendix II). The results in Figure 19 also provide an indirect check on the effectiveness of the independent methods (i.e., Equation 26 and watershed land use nutrient export coefficients) used by these reviewers to check on the reasonableness of the reported US OECD water body phosphorus loadings. The anomalies seen in both the investigator-indicated and phosphorus loading diagram-derived trophic states in Figure 19, and those seen in Figures 14 and 15 as related to the results in Figure 19, are discussed on a water body-by-water body basis in the following sections.

Based on the agreement of the investigator-indicated trophic states of the US OECD water bodies with the results indicated on the Vollenweider phosphorus loading diagram (Figure 19), and on the results of the methods used to check on the reasonableness of the reported phosphorus loadings (Figures 14 and 15), the investigator-indicated phosphorus loadings and trophic states of a majority of the US OECD water bodies appear to be reasonable. In general, they are indicative of the present trophic conditions of these water bodies. For the purposes of this report these reviewers defined a reasonable phosphorus loading to a US OECD water body as one which is within a factor of two (i.e., $\pm$ two-fold) above or below the phosphorus loadings predicted in Figures 14 and 15. There was no technical basis for choosing a factor of ± two to define a reliable phosphorus loading. A different value may be as appropriate. However, Vollenweider (1977) has indicated that the standard deviation of the relative error, considering $1/(1 + \sqrt{\tau_\omega})$ as the reference value, corresponds very well with the ± 2x assumption. A lack of agreement between the calculated and reported phosphorus loads in Figures 14 and 15 could be due either to errors on the part of the investigator in estimating nutrient loads for the lake, or to different phosphorus transport and cycling behavior in the lake's watershed and in the lake itself than is typically found for most other lakes. It should be noted that the implementation of these approaches (Figures 14 and 15) to check the reported US OECD data has caused some US OECD eutrophication study investigators to critically reexamine their nutrient load estimates, resulting in their finding errors in their original loading estimates. The methods presented in this report have been used by these reviewers to correct for these types of errors.

The failure of a particular lake or impoundment to fit the Vollenweider nutrient load-trophic state relationship may also be due to several other factors in addition to errors in phosphorus loading estimates. Particularly important would be errors in estimating hydraulic residence times, as well as personal biases of the investigators in assigning a particular trophic state classification to their water body.

It is very important to also note that a lack of fit of a particular lake to the Vollenweider total phosphorus load and mean depth/hydraulic residence time trophic state relationship does not mean that there have been errors on the part of the investigator in estimating any of these parameters. It is quite probable that even though Vollenweider and this study have found good agreement of this relationship for a wide variety of lakes and impoundments, there will be some water bodies which do not fit this relationship. This non-fitting group of lakes and impoundments would be of particular interest and significance since they would demonstrate apparently unusual phosphorus utilization. From the point of view of water quality management, it is important to clearly identify water bodies of this type so that appropriate modifications of the Vollenweider nutrient loading relationship can be made to any water quality standards that are developed by water pollution control agencies based on this relationship for these water bodies. It is important to note that the Vollenweider loading diagram is a log-log relationship. Therefore, small errors in estimating any of the parameters will not change the position of a particular water body on the diagram to any large extent. This also indicates that a large change in phosphorus loading to a water body is necessary before a significant change in trophic state can be expected.

For example, consider the possibility that the investigator-indicated phosphorus loadings to Dutch Hollow (11) were overestimated three-fold in Figure 19. If one corrected the reported phosphorus loading for this error, Dutch Hollow would still be in the eutrophic zone of the Vollenweider loading diagram. Using the same reasoning, the phosphorus loadings to Dogfish (10) could be increased four-fold, and yet Dogfish would remain in the oligotrophic zone of the Vollenweider phosphorus loading diagram. Therefore, these reviewers examined the investigator-indicated phosphorus loadings and trophic states for the possibility of an error if the reported and predicted trophic states of a given water body were not in agreement in Figure 19 and its reported and predicted phosphorus loadings were not in agreement in Figures 14 and 15.

There were only a few water bodies which showed a disagreement in one or more parameters. Lake Cayuga (6) and Sammamish (33) plot with water bodies in the eutrophic zone of the Vollenweider phosphorus loading diagram (Figure 19). Yet these two water bodies were classified as mesotrophic by Oglesby (1975) and Welch et al. (1975), respectively, on the basis of the structure and productivity of their biological communities. These investigators felt those factors were more indicative of the true trophic states of these two water bodies than were their positions on the Vollenweider phosphorus loading diagram. If the investigator-indicated trophic states of Lakes Cayuga and Sammamish are accurate, then their positions on the Vollenweider phosphorus loading diagram (Figure 19) indicate that the Vollenweider relationship between phosphorus loadings and $\bar{z}/\tau_{\omega}$ characteristics does not hold for Lakes Cayuga and Sammamish, or that the phosphorus loadings indicated by Oglesby (1975) and Welch et al. (1975), respectively, for these two water bodies may have been overestimated.

It should be mentioned here that a water body does not abruptly change in character as soon as it crosses one of the boundary lines (i.e., permissible or excessive) in the Vollenweider phosphorus loading diagram. These boundary lines were established on the basis of a subjective determination between nutrient concentration and water quality. As mentioned in an earlier section of this report, it would generally be expected that those water bodies, with a given mean depth/hydraulic residence time relationship, which have the greater vertical displacement under the permissible boundary line on the Vollenweider phosphorus loading diagram (Figure 19) would have the best water quality. Conversely, those water bodies of the greater vertical displacement above the permissible loading line would have the poorer water quality. There is a continual gradient of water quality between these two extremes, with the permissible boundary line defining a general water quality condition acceptable to the population.

The possibility of overestimation of the reported phosphorus loadings for Cayuga (6) and Sammamish (33) is consistent with the results of Figure 14 for Lake Cayuga, and with Figure 15 for Lake Sammamish. The results of Figure 14 indicate that the reported phosphorus loadings for Lake Cayuga may have been overestimated almost two-fold. Likewise, the results of Figure 15 indicate the reported phosphorus loadings for Lake Sammamish may also have been slightly overestimated. A reduction of the phosphorus loading estimates for these two water bodies to the extent indicated in Figures 14 and 15 would place them closer to the mesotrophic zone of the Vollenweider loading diagram (Figure 19), more in agreement with their investigator-indicated trophic states.

One other factor that should be considered in examination of the US OECD investigator-indicated trophic states for these two water bodies is that they were established by interpretation of classical response parameters, specifically their biological characteristics. Such interpretation is subjective in nature. When, for example, does a lake change in character from mesotrophic to eutrophic? Thus, the lack of agreement between the predicted and reported trophic states for these two water bodies could be attributed to a small error in phosphorus loading estimates, $\bar{z}/\tau_\omega$ values or the still subjective nature of trophic state classification of water bodies. Oglesby (1977) has also indicated that, in the case of Lake Cayuga, about 75 percent of the tributary total phosphorus load is adsorbed to soil particles in the tributary waters. Only about 5 percent of this adsorbed phosphorus becomes desorbable in phosphorus free aqueous solution. Thus, according to Oglesby, a significant portion of the tributary phosphorus load becomes unavailable for phytoplankton assimilation. This interpretation is consistent with Lake Cayuga's lower biological productivity in spite of a phosphorus load which places it in the eutrophic zone of the Vollenweider diagram.

Lower Lake Minnetonka-1973 (26) plots just inside the oligotrophic zone on the Vollenweider phosphorus loading diagram (Figure 19). However, Megard (1975) classified Lower Lake Minnetonka as eutrophic, changing to mesotrophic, suggesting a phosphorus loading underestimation for this water body. Sewage effluents, which was approximately 80 percent of the total phosphorus input, were diverted from Lower Lake Minnetonka in late 1971-early 1972. Yet, the eutrophic condition reported for this water body was indicative of Lower Lake Minnetonka in 1973. This situation is explainable by the fact that while the phosphorus loadings to this water body have decreased approximately 80 percent, the water body has not yet had sufficient time to shift to a new equilibrium phosphorus concentration.

Megard (1975) has indicated that Lower Lake Minnetonka appears to be slowly shifting to a mesotrophic condition, based on its mean chlorophyll concentrations and Secchi depth measurements. It is possible, unless other unusual circumstances are present, the trophic state indicated by its 1973 position in the oligotrophic-early mesotrophic zone of the Vollenweider phosphorus loading diagram (Figure 19) will be indicative of its trophic state when it has reached a new phosphorus equilibrium condition.

It is also possible that the reported 1973 phosphorus loadings for Lower Lake Minnetonka may actually have been underestimated. (note: This predicted underestimation was subsequently substantiated by Megard (1977).) Such a possibility is suggested in Figure 14 based on the reported mean phosphorus concentrations for this water body. One of the necessary parameters needed for Equation 26, which serves as the basis for Figure 14, is an accurate knowledge of the mean phosphorus concentration in the water body. If the current mean phosphorus concentration in Lower Lake Minnetonka is in a non-equilibrium condition, with respect to its phosphorus loading, because of its recent remedial treatment, the mean phosphorus concentration in Equation 26 is not justified. Its mean phosphorus concentration, and any predicted phosphorus loading based on its mean phosphorus concentration, will change with time until a new steady state condition is reached in Lower Lake Minnetonka.

No watershed land usage data was available for Lower Lake Minnetonka. Consequently, Figure 15 could not be used to check on the reasonableness of its 1973 phosphorus loading estimate.

Both Lower Lake Minnetonka and Lake Washington have undergone partial or total sewage diversion from the watershed basin. In the past, it has been common practice to relate the response of a water body which has undergone nutrient input reduction to the hydraulic residence time, or filling time (i.e., water body volume (m^3)/annual inflow volume (m^3/yr)) of the water body. However, in the case of phosphorus, such an approach does not take into consideration the aqueous chemistry of phosphorus in its role of limiting aquatic plant growth. It is more realistic to relate the rate of recovery of a water body, following nutrient input reduction, to the chemical residence time of the critical aquatic plant limiting nutrient for that water body, rather than to its hydraulic residence time. This approach in evaluating the recovery of Lake Washington and Lower Lake Minnetonka will be discussed in a following section.

AS APPLIED IN THE PHOSPHORUS RESIDENCE TIME MODEL

It is generally accepted that steady state conditions in a water body are approached exponentially in accordance with the hydraulic residence time of the water body. Assuming a lake is a completely mixed reactor subjected to continual and constant chemical influx, which only occurs through the outlet, the dynamics of a conservative substance can be described as:

$$V\, dc/dt = Qc_i - Qc \qquad (27)$$

where V = lake volume (L^3),

Q = volumetric flow rate (L^3T^{-1})

c_i = influent concentration of substance c (ML^{-3}), and

c = lake concentration of substance c (ML^{-3}).

Integrating and applying the boundary condition that $c=c_o$ at $t=o$,

$$c = c_i + (c_o - c_i)e^{-t/\tau_\omega} \tag{28}$$

where $\tau_\omega = V/Q$ = hydraulic residence time.

This latter equation shows that after a change (increase or decrease) in the incoming flux of substance c, steady state conditions are approached exponentially in accordance with the basin's hydraulic residence time. According to Rainey (1967) and Vollenweider (1969), three hydraulic residence times are required to reach 95 percent of the new steady state concentrations of substance c, following a change in the rate of supply of that substance.

However, in the case of phosphorus this approach does not consider the aqueous chemistry of phosphorus as it relates to limiting aquatic plant growth. Phosphorus is a non-conservative substance which undergoes transformations in natural waters. Accordingly, the recovery of a water body to remedial phosphorus treatment, whether it involves sewage treatment or diversion, is more accurately related to the phosphorus chemical residence time than to the hydraulic residence time. Once the residence time of the aquatic plant limiting nutrient (phosphorus or nitrogen) to a given water body is known, the rate of the water body's response to remedial treatment can be predicted if an adequate model is available.

One of the frequently-asked questions in eutrophication control programs is the rate at which the lake will come to a new equilibrium condition of water quality after altering the nutrient input. There are several deficiencies in Rainey's approach when it is applied to non-conservative substances, such as phosphorus. First, the steady-state lake concentration of phosphorus is assumed identical to the influent concentration. In reality, annual mean phosphorus concentrations are often lower than the annual input concentration of phosphorus. Second, the lake losses are assumed to occur only through the outlet. In fact, the major loss of phosphorus in lakes usually occurs as a result of sedimentation, not outflow discharge.

Accordingly, the initial equation (Equation 27) can be modified to account for these deficiencies. To account for internal losses, the expression for phosphorus (P) dynamics becomes

$$V\ dP/dt = QP_i - QP - kPV \qquad (29)$$

where K = internal loss rate constant, T^{-1}.

An assumption in this model is that the sedimentation loss is directly proportional to the mean lake phosphorus content, rather than to the phosphorus supply. One other factor that must be considered is that in stratified lakes, different water layers may contain different amounts of phosphorus due to biological, chemical and/or physical processes. An example is the summer growth period where the phosphorus concentration may only be a fraction of the whole lake concentration due to algal uptake. Thus, the outwash concentrations may be different during the summer time than during periods of lesser productivity. Accordingly, the above equation may be modified as:

$$V\ dP/dt = QP_i - Q \propto P - kVP \qquad (30)$$

where $\propto$ = dimensionless proportionality factor relating annual mean outwash or surface water phosphorus concentration to the mean annual concentration over the whole lake.

Sonzogni *et al*. (1976) have modified this model to predict changes in the phosphorus concentration as a response to nutrient input reductions based on the concept of a phosphorus residence time in natural waters. Equation 30 can be rearranged as:

$$dP + ((Q \propto + kV)/V)\ Pdt = (Q/V)\ P_i dt \qquad (31)$$

Since $V/Q = \tau_\omega$, Equation 31 can be simplified as

$$dP + (1/R_p)\ Pdt = (1/\tau_\omega)\ P_i dt \qquad (32)$$

where $R_p = V/(Q \propto + kV)$ = phosphorus residence time in lake

If $P = P_0$ at $t = 0$, Equation 32 can be integrated to produce

$$P = P_i(R_p/\tau_\omega) - (P_i(R_p/\tau_\omega) - P_0)\ e^{-t/R_p} \qquad (33)$$

The steady state phosphorus concentration is not equal to the input phosphorus concentration, but rather differs by the ratio of the phosphorus and hydraulic residence times, as

$$P\alpha = P_i(R_p/\tau_\omega) \qquad (34)$$

Thus, the time dependent solution to Equation 33 becomes

$$(P_{(t)}-P\alpha)/(P_o-P\alpha)=e^{-t/R_p} \qquad (35)$$

According to Sonzogni *et al*. (1976), if it is assumed that the water body phosphorus content was in a steady-state condition prior to remedial treatment, then one may compute the phosphorus residence time using data obtained prior to the remedial treatment. A simple approach for determining the phosphorus residence time for a water body is to divide the mean annual phosphorus content (mg P) by the annual phosphorus input to the water body (mg P/yr).

This approach was used with the US OECD water bodies. The predicted phosphorus residence times for the US OECD water bodies, based on this approach, are presented in Table 21. If a data range was reported for a water body, the mean value was used in all calculations. In addition, the inorganic nitrogen residence time has been calculated in the same manner as the total phosphorus residence time for the US OECD water bodies for which sufficient data was available. Unfortunately, while most of the US OECD investigators indicated the mean inorganic nitrogen (i.e., $NH_4^+ + NO_3^- + NO_2^-$ as N) concentrations in their reports, the organic nitrogen concentrations were not usually reported. The nitrogen residence times of the US OECD water bodies, based only on the inorganic nitrogen content, would be shorter than their actual nitrogen residence times. In addition, the relationship between the nitrogen concentrations and the nitrogen residence time would necessarily be more complex since a gaseous phase must be considered in the aqueous chemistry of nitrogen due to nitrogen fixation and denitrification reactions (Torrey and Lee, 1976; Sonzogni *et al*., 1976).

Examination of Table 21 shows that in nearly every case, the phosphorus residence time is shorter than the hydraulic residence time, usually by at least several-fold because of the environmental aqueous chemistry of phosphorus. New steady state phosphorus concentrations would be approached exponentially as a function of the phosphorus residence times (Sonzogni *et al*., 1976). As with the hydraulic residence time, 95 percent of the expected change in the water body mean phosphorus concentration following remedial treatment will be reached in a time period equal to three phosphorus residence times. Table 21 shows that, in general, for the US OECD water bodies, the oligotrophic water bodies have phosphorus residence times approaching their hydraulic residence time. The eutrophic water bodies appear to have the shortest phosphorus residence times.

Table 21. PHOSPHORUS AND NITROGEN RESIDENCE TIMES OF US OECD WATER BODIES

Water Body	Trophic State[a]	Phosphorus Mass in Water Body (mg P)	Phosphorus Loading (mg P/yr)[b]	Phosphorus Residence Time, R_p (yr)[c]	Hydraulic Residence Time, τ_ω (yr)[d]	Inorganic Nitrogen Mass in Water Body (mg N)[e]	Inorganic Nitrogen Loading (mg N/yr)[e]	Inorganic Nitrogen Residence Time, R_n (yr)[f]
Blackhawk	E	3.71×10^{8}	$1.9-2.1\times10^{9}$	0.2	0.5	3.40×10^{9}	-	-
Brownie	E	-	8.59×10^{7}	-	2.0	$< 2.73\times10^{7}$	-	-
Calhoun	E	1.91×10^{9}	1.46×10^{9}	1.3	3.6	$< 9.91\times10^{8}$	-	-
Camelot-Sherwood	E	2.94×10^{8}	$6.6-7.5\times10^{9}$	0.04	0.09-0.14	6.97×10^{9}	-	-
Canadarago	E	$2.34-2.93\times10^{9}$	6.0×10^{9}	0.4	0.6	$2.22-2.57\times10^{10}$	1.37×10^{11}	0.2
Cayuga	M	1.84×10^{11}	1.36×10^{11}	1.4	8.6	$3.4-4.68\times10^{12}$	2.46×10^{12}	1.6
Cedar	E	2.32×10^{8}	2.41×10^{8}	1.0	3.3	$< 2.31\times10^{8}$	-	-
Cox Hollow	E	1.19×10^{8}	$6.3-8.1\times10^{9}$	0.2	0.5-0.7	8.89×10^{8}	-	-
Dogfish	O	1.16×10^{7}	5.8×10^{6}	2.0	3.5	4.52×10^{8}	-	-
Dutch Hollow	E	6.63×10^{8}	$8.1-8.6\times10^{8}$	0.8	1.8	1.1×10^{9}	-	-
George	O-M	1.68×10^{10}	7.7×10^{9}	2.2	8	9.9×10^{10}	-	-
Harriet	E	7.64×10^{8}	9.94×10^{8}	0.8	2.4	$< 6.78\times10^{8}$	-	-
Isles	E	1.25×10^{8}	8.53×10^{8}	0.1	0.6	$< 6.24\times10^{7}$	-	-
Kerr Reservoir	E-M							
Roanoke Arm	-	3.71×10^{10}	6.24×10^{11}	0.06	0.2	3.46×10^{11}	-	-
Nutbush Arm	-	1.23×10^{10}	3.5×10^{10}	0.4	5.1	9.02×10^{10}	-	-
Lamb	O	1.92×10^{7}	1.21×10^{7}	1.6	2.3	8.16×10^{8}	-	-
Meander	O	1.62×10^{7}	1.08×10^{7}	1.5	2.7	8.1×10^{8}	-	-

Table 21 (continued). PHOSPHORUS AND NITROGEN RESIDENCE TIMES OF US OECD WATER BODIES

Water Body	Trophic State[a]	Phosphorus Mass in Water Body (mg P)	Phosphorus Loading (mg P/yr)[b]	Phosphorus Residence Time, R_p (yr)[c]	Hydraulic Residence Time, τ_ω (yr)[d]	Inorganic Nitrogen Mass in Water Body (mg N)[e]	Inorganic Nitrogen Loading (mg N/yr)[e]	Inorganic Nitrogen Residence Time, R_n (yr)[f]
Mendota	E	7.02×10^{10}	4.65×10^{10}	1.5	4.5	3.0×10^{11}	3.48×10^{11} (NH_4^++NO_3^--N)	0.9
Michigan (Open Water-1974)	O	6.33×10^{13}	5.8×10^{12}	11	30-100	8.28×10^{14}	7.54×10^{13}	11
Lower Lake Minnetonka								
1969	E	1.30×10^{10}	1.31×10^{10}	1.0	6.3[g]	-	-	-
1973	E→M	1.09×10^{10}	2.62×10^{9}	4.2 (7.0)[j]	6.3[g]	-	-	-
Potomac Estuary								
Upper Level	-	8.21×10^{10} -3.28×10^{11}	4.84×10^{12}	0.04	0.04	4.92×10^{11} -8.76×10^{11}	-	-
Middle Reach	-	1.07×10^{10} -8.03×10^{11}	1.68×10^{12}	0.2	0.18	1.60×10^{11} -3.53×10^{11}	-	-
Lower Reach	-	1.51×10^{11} -3.02×10^{11}	8.4×10^{11}	0.3	0.85	2.52×10^{11} $-7.56.10^{11}$	-	-
Redstone	E	7.52×10^{8}	$3.6-4.2\times10^{9}$	0.2	0.7-1.0	5.97×10^{9}	-	-
Sallie	E	1.19×10^{10}	7.95×10^{9} -2.23×10^{10}	0.8	1.1-1.8	1.49×10^{10}	-	-
Sammamish	M	1.08×10^{10}	1.4×10^{10}	0.8	1.8	6.48×10^{10}	2.6×10^{11} (NO_3^-+NO_2^--N)	0.2
Shagawa	E	3.15×10^{9}	6.44×10^{9}	0.5	0.8	8.39×10^{9}	-	-
Stewart	E	2.85×10^{6}	$1.2-2\times10^{8}$	0.02	0.08	7.41×10^{7}	-	-

Table 21 (continued). PHOSPHORUS AND NITROGEN RESIDENCE TIMES OF US OECD WATER BODIES

Water Body	Trophic State[a]	Phosphorus Mass in Water Body (mg P)	Phosphorus Loading (mg P/yr)[b]	Phosphorus Residence Time, R_p (yr)[c]	Hydraulic Residence Time, τ_ω (yr)[d]	Inorganic Nitrogen Mass in Water Body (mg N)[e]	Inorganic Nitrogen Loading (mg N/yr)[e]	Inorganic Nitrogen Residence Time, R_n (yr)[f]
Tahoe	U-O	4.70×10^{11}	2.50×10^{10}	19	700	3.13×10^{12}	-	-
East Twin								
1972	E	1.08×10^{8}	1.89×10^{8}	0.6	0.8	7.83×10^{8}	8.48×10^{9}	0.1
1973	E	1.08×10^{8}	1.35×10^{8}	0.8	0.9	1.13×10^{9}	5.21×10^{9}	0.2
1974	E	1.08×10^{8}	1.89×10^{8}	0.6	0.5	-	-	-
West Twin								
1972	E	1.77×10^{8}	1.36×10^{8}	1.3	1.6	1.17×10^{9}	5.44×10^{9}	0.2
1973	E	1.62×10^{8}	1.02×10^{8}	1.6	1.8	1.22×10^{9}	5.10×10^{9}	0.2
1974	E	1.47×10^{8}	1.02×10^{8}	1.4	1.0	-	-	-
Twin Valley	E	1.51×10^{8}	$1.06\text{-}1.25 \times 10^{9}$	0.1	0.4-0.5	8.58×10^{8}	-	-
Virginia	E	2.60×10^{7}	$2.07\text{-}2.66 \times 10^{8}$	0.1	0.9-2.8[h]	6.12×10^{7}	-	-
Waldo	U-O	4.86×10^{9}[i]	4.59×10^{8}	10	21	9.72×10^{9}[i]	-	-
Washington								
1957	E	6.97×10^{10}	1.06×10^{11}	0.6 (0.7)[j]	2.4	3.48×10^{11}	2.73×10^{11}	1.3 (1.1)[j]
1964	E	1.92×10^{11}	2.02×10^{11}	1.0 (1.0)	2.4	6.97×10^{11}	5.65×10^{11}	1.2 (0.3)
1971	M	5.23×10^{10}	3.76×10^{10}	1.4 (1.3)	2.4	5.23×10^{11}	5.60×10^{11}	0.9 (0.3)
1974	M	-	4.13×10^{10}	- (1.4)	2.4	-	-	- (0.5
Weir	M	1.21×10^{10}	3.29×10^{9}	3.7 (24.9)	4.2	1.06×10^{10}	-	- (0.2)
Wingra	E	2.35×10^{8}	1.26×10^{9}	0.2	0.4	1.04×10^{9}	7.20×10^{9}	0.1

Table 21 (continued). PHOSPHORUS AND NITROGEN RESIDENCE TIMES OF US OECD WATER BODIES

Water Body	Trophic State[a]	Phosphorus Mass in Water Body (mg P)	Phosphorus Loading (mg P/yr)[b]	Phosphorus Residence Time, R_p (yr)[c]	Hydraulic Residence Time, τ_ω (yr)[d]	Inorganic Nitrogen Mass in Water Body (mg N)[e]	Inorganic Nitrogen Loading (mg N/yr)[e]	Inorganic Nitrogen Residence Time, R_n (yr)[f]

EXPLANATION:

[a]Investigator-indicated trophic state:

E = eutrophic, M = mesotrophic, O = oligotrophic, U = ultra

[b]Based on investigator's estimates.

[c]Phosphorus residence time, R_p = annual mean total phosphorus content (mg)/annual total phosphorus input (mg/yr).

[d]Hydraulic residence time, τ_ω = water body volume (m^3)/annual inflow volume (m^3/yr).

[e]Based on investigator's estimates; includes NH_4^+ + NO_3^- + NO_2^- as N, unless otherwise indicated.

[f]Inorganic residence time, R_n = annual mean inorganic nitrogen content (mg)/annual inorganic nitrogen input (mg/yr).

[g]Hydraulic residence time of whole lake.

[h]Possible error in hydraulic residence time.

[i]Mean August value.

[j]Data in parentheses represents data received by these reviewers from the principal investigator subsequent to completion of this report. Examination of this data indicated no significant changes in the overall conclusions concerning these water bodies.

Dash (-) indicates data not available.

Lake Michigan has a hydraulic residence time ranging from 30-100 years (Piwoni et al., 1976). If it is assumed that phosphorus behaves as a conservative element in Lake Michigan it should require approximately 100-300 years for Lake Michigan to reach a new phosphorus equilibrium state following reduction of its phosphorus loading. However, the phosphorus residence time, based on the US OECD data, is approximately 10 years. Thus, the phosphorus residence time model of Sonzogni et al. (1976) predicts that it would only take approximately 30-35 years to achieve 95 percent of the expected change in the phosphorus content in Lake Michigan following a reduction in its phosphorus loading.

Megard (1977) has indicated that the quantity of phosphorus in Lower Lake Minnetonka was just beginning to move toward a new equilibrium condition in 1973 because the phosphorus load was reduced in 1971-1972, following sewage diversion from the water body. He estimated, on the basis of an adjusted phosphorus residence time (see below) that a new phosphorus equilibrium would not be reached until 1979, approximately seven years after diversion, (Megard, 1975) as compared with the 4.2 years indicated in Table 21. Prior to the sewage diversion, the phosphorus residence time was calculated to be 1.1 years, as compared to one year in Table 21. However, Megard (1977) has indicated that the predicted mean phosphorus concentration at the new equilibrium, based on a 1.1 year residence time, would only be about 14 µg/l, atypical of other lakes of the region. Consequently, he obtained a more conservative estimate of 26 µg P/l at a new equilibrium by adjusting the new phosphorus residence time upward from 1.1 to 2.0 years.

However, Megard (1977) has also noted that the 1.1 year phosphorus residence time in Lower Lake Minnetonka is based on extensive data and should be considered an accurate estimate. Since the post diversion phosphorus load is an estimate of residual influx from non-point sources, it is necessarily more tenuous than the prediversion estimate (Megard, 1977). Consequently, Megard suggests the post diversion phosphorus load estimate might be adjusted up by a factor of 1.8 (i.e., the factor used to adjust the residence time) to produce a post diversion loading of 180 mg P/m^2/yr, as compared to the 100 mg P/m^2/yr reported originally. Adjusting the load by this 1.8 factor produces the same 26 µg P/l mean concentration, at the new equilibrium, as is obtained by increasing the phosphorus residence time by the same factor (Megard, 1975). That is, the computed rate of response would still be consistent with the observed response during the first two years after diversion.

Lake Weir has a calculated phosphorus residence time of 3.7 years versus a reported value of 24.9 years (Table 21). Messer (1977) has indicated that, in addition to the mean depth, the flushing rate, or hydraulic residence time, is the principal reason for the inverse relationship between critical phosphorus load and hydraulic load. According to Messer, while this may be true for northern temperate drainage lakes which are ice-covered during part of the year, Lake Weir is a sub-tropical seepage lake located in Florida. While temperate lakes may lose 10 percent of their hydraulic load through evaporation, Lake Weir appears to lose about 83 percent of its hydraulic load due to evaporation (Messer, 1977). This heavy evaporation loss is not flushing phosphorus from the lake. Consequently, Messer suggests using the hydraulic flushing rate, exclusive of evaporation, as an estimate of the "effective flushing rate." For Lake Weir, the hydraulic residence time, exclusive of evaporation, was calculated to be about 24.9 years and indicates the increased sensitivity of Lake Weir to phosphorus inputs, relative to non-seepage water bodies.

Lake Washington provides an example of a lake which has responded to a decreased nutrient flux. Table 21 shows that, based on its hydraulic residence time, Lake Washington would require about seven to eight years to reach a new phosphorus equilibrium condition. However, the response of the lake to nutrient reduction has been both prompt and sensitive (Edmondson 1970b, 1972). The lake was considered highly eutrophic in 1964. Yet, by 1971, following completion of the sewage diversion project in the late 1960's, the lake was re-classified as mesotrophic by Edmondson (1969, 1970b). The phosphorus residence time was calculated as 0.5 years in Table 21. Consequently, one would expect a 95 percent recovery of the lake in one to two years following the sewage diversion. This situation was in fact seen in Lake Washington following completion of sewage diversion in the late 1960's (Edmondson, 1970b; Sonzogni, _et al_., 1976).

Megard (1971) compared the actual rate at which the phosphorus concentration in Lake Washington decreased, following sewage diversion, with the phosphorus concentration predicted from the phosphorus residence time model. He found the observed rates of decrease paralleled the predicted rates, and the measured phosphorus concentrations were similar to the predicted phosphorus concentrations. Based on these results, Lake Washington provides a successful test of the phosphorus residence time model as an approach to assessing the rate of recovery of a water body following phosphorus input reduction.

AS APPLIED IN VOLLENWEIDER EQUATION FOR CRITICAL PHOSPHORUS LOADING

In addition to his phosphorus loading diagrams, Vollenweider (1976a) had derived several equations for calculating the critical phosphorus loading levels and expected trophic states for lakes and impoundments. As indicated earlier, Equation 19 expresses a generalized relationship which can be used to determine critical phosphorus loads for lakes and impoundments, based on their mean depth and hydraulic residence time characteristics.

According to Vollenweider (1976a), assuming steady state conditions, water bodies which receive phosphorus loadings below the critical level defined by Equation 19 would be expected to be in an oligotrophic condition. Conversely, water bodies whose phosphorus loadings were more than twice the critical loading level would be expected to be eutrophic. A water body with phosphorus loadings between these two limits would be mesotrophic.

Equation 19 was used by these reviewers to check the reported phosphorus loading levels and trophic states for the US OECD water bodies. The pertinent data for the US OECD water bodies is presented in Table 22. If a data range was reported for a water body, the mean value was used in all calculations. The last column in Table 22 indicates the approximate factor by which the investigator-indicated phosphorus loading exceeds or falls short of the predicted critical phosphorus loading level predicted by Equation 19. For example, Lake Canadargo's reported phosphorus loading is approximately 3.5 times greater than its calculated critical phosphorus loading level. Conversely, Lake Waldo could adsorb a phosphorus loading increase of over 5.6-fold and still retain its oligotrophic character, according to Equation 19. Lake Washington, having a reported phosphorus loading between one and two times the predicted critical loading, would be classified as mesotrophic in 1974 on the basis of Equation 19.

Overall, the results of Table 22 are essentially identical to those illustrated in the Vollenweider phosphorus loading diagram (Figure 19). As the investigator-indicated trophic conditions are in good agreement with the trophic states indicated in Table 22, this lends further support to the use of these two methodologies for determining the critical phosphorus loads to water bodies in a variety of trophic conditions.

COMPARISON OF RESULTS

Before the OECD eutrophication study data can be evaluated with the Vollenweider phosphorus loading criteria, any discrepancies between the predicted and reported phosphorus loading and trophic conditions of the US OECD water bodies should be explained. This was attempted in previous sections in this report. It is also necessary to try to explain why some US OECD water bodies appear to plot accurately on the Vollenweider phosphorus loading diagram, based on their reported phosphorus loading and mean depth/hydraulic residence time characteristics and tro-

Table 22. US OECD DATA USED IN VOLLENWEIDER'S CRITICAL PHOSPHORUS LOADING EQUATION

Water Body	Hydraulic Loading, q_s (m/yr)[a]	Calculated Critical Phosphorus Loading, $L_c(P)$ (mg P/m^2/yr)[b]	Calculated Trophic State[c]	Investigator-Indicated Phosphorus Loading (mg P/m^2/yr)[d]	Investigator-Indicated Trophic State[c,d]	Factor Relating Investigator-Indicated Loading to Calculated Loading[e]
Blackhawk	9.8	167	E	2130-2320	E	+13 to +14
Brownie	3.4	82	E	1180	E	+14.4
Calhoun	2.94	85	E	860	E	+10.1
Camelot-Sherwood	21.4-33.3	294-433	E	2350-2680	E	+5.4 to +9.1
Canadarago	12.8	227	E	800	E	+ 3.5
Cayuga	6.28	247	E	800	M	+ 3.2
Cedar	1.85	52	E	350	E	+ 6.7
Cox Hollow	5.4-7.6	93-130	U-E	1620-2080	E	+12.5 to +21.0
Dogfish	1.14	33	O	20	O	- 1.6
Dutch Hollow	1.67	39	E	950-1010	E	+24 to +25
George	2.25	86	O	70	O-M	- 1.2
Harriet	3.67	94	E	710	E	+ 7.6
Isles	4.5	80	U-E	2030	E	+ 25
Kerr Reservoir						
Roanoke Arm	51.5	745	E	5200	E-M	+ 7.0
Nutbush Arm	1.61	52	U-E	700	E-M	+ 14

Table 22 (continued). US OECD DATA USED IN VOLLENWEIDER'S CRITICAL PHOSPHORUS LOADING EQUATION

Water Body	Hydraulic Loading, q_s (m/yr)[a]	Calculated Critical Phosphorus Loading, $L_c(P)$ (mg P/m^2/yr)[b]	Calculated Trophic State[c]	Investigator-Indicated Phosphorus Loading (mg P/m^2/yr)[d]	Investigator-Indicated Trophic State[c,d]	Factor Relating Investigator-Indicated Loading to Calculated Loading[e]
Lamb	1.74	44	O	30	O	- 1.5
Meander	1.85	49	O	30	O	- 1.6
Mendota	2.67	83	U-E	1200	E	+ 14
Michigan (open waters)						
1971 (τ_ω = 30 yr)	2.8	181	O	140	O	- 1.3
1974 (τ_ω = 30 yr)	2.8	181	O	100	O	- 1.8
1971 (τ_ω = 100 yr)	0.84	92	O	140	O	+ 1.5
1974 (τ_ω = 100 yr)	0.84	92	O	100	O	+ 1.1
Lower Lake Minnetonka						
1969	1.32[f]	46	E	500	E	+ 11
1973	1.32[f]	46	E	100 (180)[g]	E-M	+ 2.2 (+3.9)[g]
Potomac Estuary						
Upper Reach	120	1440	E	85000	U-E	+59
Middle Reach	28.3	403	E	8000	U-E	+20
Lower Reach	8.47	163	E	1200	U-E	+ 7.4
Redstone	4.3-6.1	86-112	E	1440-1680	E	+13 to +20
Sallie	3.56-5.82	83-119	E	1500-4200	E	+13 to +51
Sammamish	10	234	E	700	M	+ 3.0
Shagawa	7.12	135	E	700	E	+ 5.2

Table 22 (continued). US OECD DATA USED IN VOLLENWEIDER'S CRITICAL PHOSPHORUS LOADING EQUATION

Water Body	Hydraulic Loading, q_s (m/yr)[a]	Calculated Critical Phosphorus Loading, L_c(P) (mg P/m^2/yr)[b]	Calculated Trophic State[c]	Investigator-Indicated Phosphorus Loading (mg P/m^2/yr)[d]	Investigator-Indicated Trophic State[c,d]	Factor Relating Investigator-Indicated Loading to Calculated Loading[e]
Stewart	23.8	305	E	4820-8050	E	+16 to +26
Tahoe	0.45	124	O	50	U-O	- 2.5
East Twin						
1972	6.25	118	E	700 (700)[g]	E	+ 5.9 (+5.9)[g]
1973	5.56	108	E	500 (500)	E	+ 4.6 (+4.6)
1974	10	171	E	700 (800)	E	+ 4.1 (+4.7)
West Twin						
1972	2.71	61	E	400 (400)	E	+ 6.5 (+6.5)
1973	2.41	56	E	300 (200)	E	+ 5.3 (+3.6)
1974	4.34	87	M	300 (300)	E	+ 3.4 (+3.4)
Twin Valley	7.6-9.5	130-155	E	1740-2050	E	+11 to +16
Virginia	0.6-0.9	16- 37	U-E	1150-1480	E	+31 to +92
Waldo	1.71	95	O	17	U-O	- 5.6
Washington						
1957	13.8	351	E	1200	E	+ 3.4
1964	13.8	351	E	2300	E	+ 6.5
1971	13.8	351	M	430	M	+ 1.2
1974	13.8	351	M	470	M	+ 1.3

Table 22 (continued). US OECD DATA USED IN VOLLENWEIDER'S CRITICAL PHOSPHORUS LOADING EQUATION

Water Body	Hydraulic Loading, q_s (m/yr)[a]	Calculated Critical Phosphorus Loading, $L_c(p)$ (mg P/m^2/yr)[b]	Calculated Trophic State[c]	Investigator-Indicated Phosphorus Loading (mg P/m^2/yr)[d]	Investigator-Indicated Trophic State[c,d]	Factor Relating Investigator-Indicated Loading to Calculated Loading[e]
Weir	1.5	46	M	140	M	+ 3.0
Wingra	6	98	E	900	E	+ 9.2

EXPLANATION:

[a]Hydraulic loading, q_s = mean depth, $\bar{z}$/hydraulic residence time, τ_ω.

[b]Based on Equation 19.

[c]E = eutrophic, M = mesotrophic, O = oligotrophic, U = ultra

[d]Based on investigator's estimates.

[e]Factor by which investigator-indicated loading exceeds (+) or falls short (-) of the critical phosphorus loading predicted by Equation 19.

[f]Hydraulic residence time for whole lake.

[g]All data in parentheses represent data received by these reviewers from the principal investigators subsequent to completion of this report. Examination of the data indicates no significant changes in the conclusions concerning these water bodies.

phic states, even though other relationships (Figures 14 or 15) indicate that the reported phosphorus loadings may be in error.

This may be partially because the Vollenweider phosphorus loading diagram is a log-log graph. This type of graph allows the values of one or both parameters being plotted to change considerably without a proportionally large change occurring in its position on the graph. As a result, the reported phosphorus loadings for many US OECD water bodies can be corrected for possible over or underestimations without altering their trophic state categorizations on the Vollenweider phosphorus loading diagram (Figure 19). The only exceptions are those water bodies which plotted near the permissible or excessive boundary lines.

Discrepancies between Vollenweider Phosphorus Loading Diagram and Vollenweider Mean Phosphorus/Influent Phosphorus And Hydraulic Residence Time Diagram

Figure 14 indicates that the reported phosphorus loading for several of the US OECD water bodies may have been under or overestimated. Those water bodies whose reported phosphorus loadings may be underestimated include Lower Lake Minnetonka-1973 (26), East Twin Lake-1974 (41), West Twin Lake-1973 and 1974 (44 and 45, respectively), Lake Waldo (48), Lake Weir (53) and the Upper Reach of the Potomac Estuary (28). Conversely, the phosphorus loadings to Lakes Isles (14), the Roanoke and Nutbush Arms of the Kerr Reservoir (16 and 17, respectively), Lake Stewart (35) and Lake Virginia (47) may have been overestimated.

Figure 15, based on watershed land usage patterns and phosphorus export coefficients, indicates the phosphorus loading estimates to Lake Dogfish (10), Lake Lamb (19), Lake Meander (22), Lake Sallie (32), Lake Tahoe (36), Lake Waldo (48) and Lake Weir (53) may have been underestimated.

Lake Waldo--

Figure 14 indicates that phosphorus loadings to Lake Waldo (48) may have been underestimated by three-fold. Waldo, which is classified as ultra-oligotrophic by Powers *et al*. (1975) falls in the ultra-oligotrophic zone of the Vollenweider phosphorus loading diagram (Figure 19). If its phosphorus loading estimates were corrected to the degree indicated in Figure 14, Lake Waldo would plot much closer to the mesotrophic zone. However, its reported nutrient and chlorophyll concentrations, primary productivity and other classical trophic state indicators indicate that Waldo is ultra-oligotrophic. It is classed among the most pristine lakes in the United States. Thus, it would appear that the phosphorus loading underestimation indicated in Figure 14 may be in error, and that the reported phosphorus loading estimate is correct.

There are several possible reasons for the disagreement between the results of Figure 14 and of the Vollenweider phosphorus loading diagram (Figure 19). The relationship expressed in Figure 14 (Equation 26) is based partly on the annual mean phosphorus concentration. Thus, use of this relationship as a check on the phosphorus loading to Lake Waldo requires an accurate knowledge of its annual mean phosphorus concentration. However, according to Powers et al. (1975), the mean phosphorus concentrations reported for Lake Waldo were determined from an annual visit to Lake Waldo in August or September from 1969 to 1974. Thus, the reported mean phosphorus concentration was the August mean value, rather than the annual mean value, and does not necessarily reflect variations in the mean phosphorus concentrations over the annual cycle. It may not be appropriate to apply the reported growing season mean phosphorus concentration for Lake Waldo to Equation 26 to check on its reported phosphorus loading. Therefore, the phosphorus loading underestimation for Lake Waldo in Figure 14 may be incorrect.

It should also be mentioned that Figure 14 is based on a relationship derived for phosphorus-limited water bodies. It is not clear that phosphorus limits algal growth in Lake Waldo (Powers et al., 1972; Miller et al., 1974).

It is possible that the reported phosphorus loading to Waldo may be in error to some degree. The phosphorus loadings were not measured directly. Rather they were based on the results of four indirect methods (Powers et al., 1975). The mean phosphorus loading was obtained by averaging the results of these four methods. However, the results of these four methods differ by nearly three-fold. An average phosphorus loading based on these methods would incorporate any errors from each method into the final value. In addition, while Powers et al. (1975) considered the phosphorus input from precipitation and fallout in their phosphorus loading estimate, they did not include the phosphorus contribution from dry fallout (Table 9). According to Kluesener (1972), Sonzogni and Lee (1974), Murphy (1974) and Murphy and Doskey (1975), dry fallout can contribute substantial quantities of phosphorus to water bodies. Kluesener (1972) reported dry fallout contributed about three times as much total phosphorus and twice as much total nitrogen to Lake Wingra than did precipitation. Murphy (1974) reported that dry fallout contributes up to 18 percent of the present phosphorus loading to Lake Michigan, and that about half of the dry fallout loading is in the form of orthophosphate, the form most readily available for algal growth. Thus, this magnitude of phosphorus input could constitute a significant fraction of the total phosphorus input to oligotrophic water bodies, which do not ordinarily have any major point-source inputs.

Lake Waldo is still in a pristine state, based on its present limnological characteristics. The phosphorus loading could be increased about five-fold, according to both Figure 19 and Table 22, without altering its trophic state association in the oligotrophic category. However, such an increase in phosphorus loading would imply a significant decrease in water quality in Lake Waldo. Its relatively deep mean depth and long hydraulic residence time, compared to the other US OECD water bodies, implies a relatively slight increase in phosphorus loading to Waldo could alter its trophic status. This view is shared by the US OECD investigator for Lake Waldo.

Lake Weir--

The phosphorus loading anomaly in Figure 14 concerning Lake Weir may be more complicated in nature. Lake Weir is atypical in several respects to the other US OECD water bodies. It is a seepage lake with no natural tributary or point-source inputs of water or phosphorus. Rather, it receives its phosphorus solely from groundwater seepage into the lake, from land runoff directly into the lake and from atmospheric sources (i.e., precipitation and dry fallout) directly onto its surface. Also, it is one of only two US OECD water bodies (Figure 4) located in a sub-tropical (i.e., warm water) setting. According to Brezonik and Messer (1975), the application of relationships which were derived in temperate zones to an area of high permeable sands, high soil temperature, unique geology and sub-tropical climate, as is found in the Lake Weir watershed, is questionable. It is possible the phosphorus loading-algal response relationships in the southern and southwestern US warm-water lakes and impoundments are different from those found in north temperate-cold water bodies. This should be remembered in examination of the phosphorus loading and trophic characterization data for Lake Weir.

Figures 14 and 15 indicate the phosphorus loadings to Lake Weir may have been underestimated by a factor of three. Table 22 also indicates the possibility of a phosphorus loading underestimation. However, Lake Weir plots in the mesotrophic zone of the Vollenweider phosphorus loading diagram, in agreement with the trophic condition reported by the investigators (Brezonik and Messer, 1975). A mesotrophic state is consistent with the results expressed in Table 22 for Lake Weir.

If the phosphorus loading estimates were corrected for the three fold underestimation indicated in Figures 14 and 15, Weir would plot in the eutrophic zone of the Vollenweider phosphorus loading diagram (Figure 19). However, Brezonik and Messer (1975) have indicated that while the concentrations of nitrogen and phosphorus are high throughout the water column and exceed Sawyer's (1947) critical concentrations at all times of the year, primary productivity in Lake Weir is low to moderate and nuisance conditions do not occur. Further, although macrophytes are common in Lake Weir, floating mats or nuisance growths of macrophytes are not found. Brezonik and Messer also indicated that generally good water quality is found in Lake Weir. These indications suggest the degree of phosphorus loading underestimation indicated for Lake Weir in Figure 14 may be in error.

Another possible reason for the disagreement between Figures 14 and 19 may result from a fundamental difference in the phosphorus loading-algal response relationships in temperate and sub-tropical systems. It is possible that both the reported phosphorus loading and trophic state of Lake Weir are correct, and that what is actually anomalous is the interpretation of the nutrient loading-algal response relationship in water bodies in subtropical environments. A phosphorus loading which would place a temperate water body in the mesotrophic zone of the Vollenweider loading diagram may produce trophic conditions in a water body (with the same mean depth/hydraulic residence time characteristics) in the sub-tropical setting of Florida which would be interpreted by most investigators as eutrophic. Brezonik et al. (1969) have presented some basic differences between northern US temperate lakes and lakes in north central Florida. Although the reported and predicted trophic conditions for Lake Weir are in agreement in Figure 19, additional research on the nutrient loading-algal response relationships in warm-water bodies may still be necessary to determine whether the Vollenweider phosphorus loading diagram is applicable in its present form, or whether the permissible and excessive boundary loading lines may have to be modified to fit different nutrient loading-algal response relationships in warm-water lakes and impoundments.

Lower Lake Minnetonka--

The phosphorus loading to Lower Lake Minnetonka-1973 (26) is indicated as possibly being underestimated about two-fold in Figure 14. Lower Lake Minnetonka plots at the early mesotrophic-late oligotrophic boundary area of the Vollenweider phosphorus loading diagram (Figure 19), although Megard (1975) has classified Minnetonka as eutrophic. Minnetonka has undergone sewage effluent diversion, completed in early 1972, reducing the annual phosphorus influx almost 80 percent. Since that time, according to Megard (1975), a decreasing mean phosphorus concentration and relative integral photosynthetic rate indicates Lower Lake Minnetonka to be changing from a eutrophic to a mesotrophic condition. This is in agreement with the results of Table 22. However, the inappropriate use of a non-equilibrium water body mean phosphorus concentration for predicting phosphorus loading is likely the reason for the loading underestimation indicated in Figure 14. This was discussed in relation with the phosphorus residence time in a previous section of this report.

Twin Lakes-1973 and 1974--

East Twin Lake-1974 (41) and West Twin Lake-1973 and 1974 (44 and 45, respectively) are indicated in Figure 14 as possibly having phosphorus loading underestimations between two and three-fold. Based on their plankton characteristics, both East Twin Lake and West Twin Lake are currently in a eutrophic condition, according to Cooke *et al.* (1975). These observations are consistent with the trophic character for these water bodies predicted in Table 22 and with the Vollenweider phosphorus loading diagram (Figure 19). This suggests the phosphorus loading underestimation indicated in Figure 14 may be in error.

As with Lower Lake Minnetonka, the reason for the Twin Lake's phosphorus loading underestimation indicated in Figure 14 is likely related to the non-equilibrium mean phosphorus concentrations of these water bodies. Sewage was diverted from the Twin Lakes during 1972 to a package plant which discharges away from the watershed. Thus, the relationship expressed in Figure 14, based partly on the mean phosphorus concentration, is likely to produce erroneous results.

The phosphorus residence time for Lower Lake Minnetonka is about four years (Table 21) while that of East Twin and West Twin is about 1 and 1.5 years, respectively. Thus, Lower Lake Minnetonka should reach a new steady-state mean phosphorus concentration in about 10 to 12 years. East Twin Lake and West Twin Lake should reach their equilibrium states in about three and five years, respectively. Thus, while their phosphorus loadings can be reduced rapidly to substantially lower levels by remedial treatments, it will take a longer period of time for these water bodies to reach new equilibrium mean phosphorus concentrations and trophic conditions. Of the three water bodies, East Twin Lake appears to be closest to a new equilibrium phosphorus concentration, based on its phosphorus residence time. This is consistent with its position on the Vollenweider phosphorus loading diagram (Figure 19) and with the results of Figure 14.

One point that should be mentioned here is that, while the Vollenweider model (Figure 19) appears to accurately predict the degree of fertility of water bodies as described by their plankton productivity characteristics, it does not address the problem of estimation of the degree of fertility expressed in macrophyte growth. The Twin Lakes have an extensive littoral area and approximately half of their primary productivity is in the form of macrophyte growth. According to Cooke *et al.* (1975), the Twin Lakes are of poorer water quality, from the point of view of the recreational user, than is indicated by the early eutrophic characterization given them by the Vollenweider phosphorus loading diagram. The Vollenweider model is based primarily on plankton characteristics, and may not be applicable in its present form to water bodies with extensive macrophyte problems such as are found in the Twin Lakes and several other US OECD water bodies, or to turbid waters as found in some Texas lakes and impoundments (Lee, 1974b).

Potomac Estuary and Lake of the Isles--

The Upper Reach of the Potomac Estuary (28) is indicated in Figure 14 to have phosphorus loading underestimations between two and three fold. The Potomac Estuary is indicated by Jaworski (1975) and on the Vollenweider phosphorus loading diagram (Figure 19) as being highly eutrophic. Table 22 also indicates that the phosphorus loads to all Reaches of the Potomac Estuary are all many-fold above the permissible loading levels. Lake of the Isles (14) is indicated in Figure 14 as having a possible phosphorus loading overestimation of about two fold. This water body is characterized by Shapiro (1975a) and on the Vollenweider phosphorus loading diagram as being highly eutrophic.

As mentioned earlier, the relationship expressed in Figure 14 requires accurate knowledge of the annual mean phosphorus concentration in the water body. The reported mean phosphorus concentrations for the Potomac Estuary and Lake of the Isles were the mean summer value and the mean summer surface value, respectively, rather than the annual mean values of these water bodies. Because these water bodies are highly eutrophic, the mean phosphorus concentration during the summer months will likely vary cyclically as a function of algal blooms and die-offs. As a result, the measured mean phosphorus concentration would be a function of when the water body was sampled. Thus, the use of the summer mean phosphorus concentration in the relationship expressed in Figure 14 as a check on the phosphorus loading is probably not valid for these water bodies.

There are several other eutrophic US OECD water bodies (i.e., Brownie, Calhoun, Cedar, Harriet) for which only the mean summer phosphorus concentration was reported, yet whose phosphorus loadings appear reasonable in Figure 14. This may be coincidental as a function of when these water bodies were sampled for their mean phosphorus concentrations. These findings are consistent with the results of Figure 15, which is not based on mean phosphorus concentrations, and which indicates the phosphorus loadings to the Potomac Estuary and Lake of the Isles to be reasonable. One additional factor to consider in examination of the Potomac Estuary data is that it has typical estuarine water circulation patterns. These circulation patterns would likely alter the nutrient loading-algal response relationships which are dependent on hydraulic residence time.

Lake Stewart, Lake Virginia and Twin Valley Lake--

The phosphorus loadings for Twin Valley Lake (46), Lake Stewart (35) and Lake Virginia (47) are indicated in Figure 14 as being overestimated by approximately two, three and four-fold respectively. These water bodies are Wisconsin impoundments with shallow mean depths and short hydraulic residence times. According to Piwoni and Lee (1975) and their position on the Vollenweider phosphorus loading diagram (Figure 19), these water bodies are highly eutrophic.

The phosphorus loading underestimation indicated in Figure 14 for Lake Virginia may be due to an error in the calculation of the hydraulic residence time (i.e., water body volume (m^3)/annual inflow volume (m^3/yr)). With a mean depth of 1.7 m, and a mean hydraulic residence time of 1.8 years, the resultant hydraulic loading, q_s ($\bar{z}/\tau_\omega$), calculates to be 0.9 m/yr. This value is unrealistically small for Lake Virginia's watershed. The meteoric discharge rate is a measure of the rate at which water is supplied to the water body from the watershed. According to Vollenweider and Dillon (1974; Vollenweider, 1976b), the relationship is expressed as

$$MDR = (q_s\ (A_o/A_d)) \qquad (36)$$

where MDR = meteoric discharge rate (m/yr),

q_s = hydraulic loading = $\bar{z}/\tau_\omega$ (m/yr),

$\bar{z}$ = mean depth (m),

τ_ω = hydraulic residence time (yr),

A_o = water body surface area (m^2), and

A_d = watershed area (m^2).

For Lake Virginia, MDR = (0.9 m/yr) ($1.8 \times 10^5\ m^2/6.5 \times 10^6\ m^2$) = 0.02 m/yr. This low meteoric discharge rate is unlikely for the Lake Virginia watershed area. The nearby Dutch Hollow Lake and Lake Redstone have meteoric discharge rates of 0.22 m/yr and 0.35 m/yr, respectively. Since the mean depth, watershed area and water body surface area appear to be correct for Lake Virginia, this suggests the hydraulic residence time may be in error, probably overestimated by a factor of ten. If the hydraulic residence time was changed from 1.8 to 0.18 years, the value for $[P]/\overline{[P]}$ in Figure 14 would change from 0.06 to

0.6, and the value for $1/(1 + \sqrt{\tau_\omega})$ would change from 0.4 to 0.7 (Table 15). These new values plotted into Figure 14 would place Lake Virginia in a position corresponding to less than a two-fold phosphorus loading overestimation, indicating that the phosphorus loading estimate for Lake Virginia is reasonable.

Piwoni and Lee (1975) have indicated that the values reported for Lake Virginia are highly uncertain because this water body is a seepage lake and may behave quite differently from a water body with a base flow surface input. They have also indicated that the phosphorus loading estimates may be high because of the very sandy soils in Lake Virginia's watershed, which would reduce overland transport of phosphorus. This would result in an indication of a possible phosphorus loading overestimation, particularly since the nutrient loadings to Lake Virginia were estimated from watershed nutrient export coefficients (Piwoni and Lee, 1975). There is also a possibility that the incoming phosphorus to Lake Virginia may be short-circuited out of the lake during high flow periods. This would also produce a misleading estimate of the phosphorus loadings based on Equation 26.

The possible phosphorus loading overestimations for Lake Stewart and Twin Valley Lake cannot be resolved in the same manner. Their hydraulic residence times appear reasonable, relative to the other impoundments in the region. If Figure 14 is incorrect such that the phosphorus loading estimates for Lake Stewart and Twin Valley Lake are reasonable, then according to Vollenweider (1976a; 1975d) the mean phosphorus concentration in these water bodies is lower than would be expected for the reported phosphorus loadings. This indicates that the sedimentation rate in these water bodies is statistically above average. Such a situation currently exists in Lake Erie (Vollenweider, 1975d). Whether this also occurs in Lake Stewart and Twin Valley Lake is unknown.

Another factor which may have to be considered is that the reported mean phosphorus concentration in these two water bodies is the average of the mean summer and mean winter values. It is not known whether a mean value derived from continuous measurements over the annual cycle would differ significantly from a mean value derived from the summer and winter average value in these two water bodies. A large difference in the value of the mean phosphorus concentrations measured by these two methods may significantly alter the indicated phosphorus loading overestimation for Twin Valley Lake and Lake Stewart in Figure 14. However, it should also be noted that the same procedure was employed by Piwoni and Lee (1975) for other US OECD impoundments in the same region and Figure 14 indicates the phosphorus loading estimates for these other impoundments to be reasonable. A factor which may influence the phosphorus in Lake Stewart compared to the other lakes is that a potentially significant part of

Lake Stewart has extensive macrophyte growth which would tend to alter the cycling of phosphorus in the lake. Therefore, the phosphorus loading overestimation indicated for Lake Stewart in Figure 14 may be incorrect.

Kerr Reservoir--

Figure 14 indicates the phosphorus loading estimates for both the Roanoke and Nutbush Arms of the Kerr Reservoir (16 and 17, respectively) may be overestimated between two and four-fold. The two arms of the Kerr Reservoir have been treated separately by Weiss and Moore (1975) because they differ significantly in their morphometric, hydrologic and limnologic characteristics. In both arms of the reservoir, there is a changing magnitude in nearly all water quality parameters as one moves from the upstream end of the arm toward the dam. In general, the nutrient and chlorophyll concentrations and associated productivity parameters decrease as one approaches the dam, indicating a relative increase in water quality in the direction of the dam. Weiss (1977) indicated this shift in water quality illustrates that the sedimentation characteristics of the upper arms of the Kerr Reservoir, and probably other river systems impoundments, have a marked impact on reduction of the phosphorus entering these water bodies. The results would be a lower net phosphorus concentration in the upper arm than expected (this was discussed earlier in relation to the inorganic nitrogen:soluble orthophosphate ratio in the Kerr Reservoir; see Tables 9 and 10). When this lower phosphorus concentration was inserted into Equation 25 the result was the predicted underestimation of phosphorus load indicated in Figure 14. Weiss (1977) noted that this interpretation was substantiated by Table 18, in which the phosphorus load prediction is based on watershed phosphorus export coefficients.

The flushing rate is believed to be the major controlling variable in establishing the relative degree of fertility and behavior differences in the two arms. According to Weiss and Moore (1975) the hydraulic residence time is approximately 70 days in the Roanoke Arm and approximately 1800 days in the Nutbush Arm. These computations are based on inflow water volume and do not consider exchange of water between the main body of the lake and the arms. The actual hydraulic residence time of the water in each arm would likely be less than the indicated amount by a factor somewhat proportional to water exchange between various parts of the lake. However, Weiss (1977) has indicated that the main flow of water through the Kerr Reservoir is down the Roanoke Arm and into the major basin above the dam. The hydraulic load down the Roanoke Arm is so much faster than the flow from the Nutbush Arm that exchange of water between the two arms is inconsequential. Weiss has indicated that this is substantiated by the fact that the phosphorus concentration at the end of the Nutbush and Roanoke Arms, where they both enter the main basin, are approximately the same, suggesting that interchange effects are negligible. The high correlation of growth parameters with the hydraulic residence time indicates the importance of this factor in establishing the relative degree of fertility of the two arms.

The two arms of the Kerr Reservoir are described as eutrophic-mesotrophic by Weiss and Moore (1975) and plot in the eutrophic zone on the Vollenweider phosphorus loading diagram (Figure 19). The two arms would still remain in the eutrophic zone of the Vollenweider loading diagram if their phosphorus loading estimates were reduced by the degree indicated in Figure 14. However, they would be closer to the excessive loading boundary line. Unfortunately, watershed land usage data was available only for the whole watershed, not for the subwatersheds of the two arms. Since the amount of mixing between the two arms could not be estimated, it was not possible to use Figure 15 to check on the reported phosphorus loadings. However, Table 22 indicates that the phosphorus loadings are many fold above the permissible level. While it is not unequivocal, this implies the phosphorus loading overestimation indicated in Figure 14 for the two arms of the Kerr Reservoir may be incorrect.

Discrepancies Between Vollenweider Phosphorus Loading Diagram and Watershed Phosphorus Export Coefficient Calculations

Dogfish Lake, Lamb Lake and Meander Lake--

Figure 15 indicates the phosphorus loadings for Lakes Dogfish (10), Lamb (19) and Meander (21) are approximately five-fold underestimated. Contrastingly, Figure 14 indicates their phosphorus loadings are reasonable. The results of Table 22 are consistent with the phosphorus loading underestimation indicated in Figure 15. Thus, it would appear that the reported phosphorus loadings and the ultra-oligotrophic conditions of Dogfish, Lamb and Meander predicted in Figure 19 may be in error. The low chlorophyll level in these water bodies indicates them to be in relatively unproductive states. However, according to Table 22, they are not in the ultra-oligotrophic state indicated by their large vertical distance below the permissible loading line on the Vollenweider phosphorus loading diagram (Figure 19). Based on their phosphorus and hydraulic loadings, these three water bodies plot in the same general area of the Vollenweider phosphorus loading diagram (Figure 19) as does Lake Waldo, implying that they exhibit about the same relative degree of oligotrophy as does pristine Lake Waldo. However, their water quality does not support the view that they are relatively as oligotrophic as Lake Waldo. The reported mean phosphorus and nitrogen concentrations are all higher in Lakes Dogfish, Lamb and Meander than those reported for Lake Waldo. Further, the mean chlorophyll concentrations are also considerably higher in Dogfish, Lamb and Meander than in Waldo, in some instances by an order of magnitude or greater. Secchi depth is also considerably greater in Waldo than in Dogfish, Lamb and Meander. However, these three water bodies are reported to have high humic color and, therefore, possibly have reduced light penetration. Consequently, comparison of Secchi depth measurements would not yield reliable information concerning the degree of oligotrophy in Dogfish, Lamb and Meander relative to Waldo. It should also be mentioned that the higher chlorophyll concentration in Dogfish, Lamb and Meander than that found in Waldo implies the color of the water is not reducing the primary production in these three water bodies to any great extent relative to Waldo.

In general, the results of Figure 15, Table 22 and the reported water quality data indicate that the reported phosphorus loadings for Lakes Dogfish, Lamb and Meander may have been underestimated, though perhaps not to the extent indicated in Figure 15. Consequently, their position on the Vollenweider phosphorus loading diagram may have to be adjusted accordingly so as to produce an accurate representation of the relative trophic states of these three water bodies.

Lake Tahoe--

Figure 15 indicates the phosphorus loading to Lake Tahoe (36) may have been overestimated by a factor of four. However, Lake Tahoe appears to be nitrogen-limited with respect to aquatic plant nutrient requirements (Table 9). As the Vollenweider phosphorus loading diagram was developed for phosphorus-limited water bodies, attempting to categorize its trophic condition based solely on its trophic state association in the Vollenweider phosphorus loading diagram may not be a valid procedure. Therefore, Lake Tahoe's nutrient loading-trophic response relationship will be examined further in an analysis of the US OECD water body nitrogen-loading estimates in a subsequent section. It should be noted that Schindler (1977) has recently indicated there appears to exist a very precise relationship between the total phosphorus concentration in a water body and the standing crop of phytoplankton, even in water bodies whose low N:P ratios should favor nitrogen limitation. This suggests that natural mechanisms may compensate for deficiencies of nitrogen in many water bodies.

Lake Sallie--

Figure 15 indicates Lake Sallie's (32) phosphorus loadings may have been underestimated between two to seven fold. The same trend is noted in Figure 14. Lake Sallie possesses one of the highest ratios of watershed area to water body surface areas of all the US OECD water bodies. Thus, its phosphorus loading is very high when it is calculated with watershed land use phosphorus coefficients. Lake Sallie plots in the ultra-eutrophic zone. However, Neel (1975) characterizes Lake Sallie as being in a late mesotrophic-early eutrophic state, suggesting the high degree of fertility indicated in Figure 19 may be in error. According to Neel, the atmospheric input of phosphorus from dry fallout was not considered in the phosphorus loading estimates. Therefore, it is possible that Lake Sallie's phosphorus loadings are underestimated to some degree. Table 22 also indicates that Lake Sallie may be more fertile than the investigator-indicated late mesotrophic-early eutrophic condition.

However, one other factor that must be considered is that the water quality problems associated with excessive nutrients in Lake Sallie are manifested to a major extent in the growth of attached macrophytes. As discussed in earlier sections of this report, the excessive and permissible loading lines on the Vollenweider phosphorus loading diagram (Figure 19) are based primarily on planktonic algal problems and may not be applicable to water bodies such as Lake Sallie which possess extensive beds of macrophytes. The relatively high phosphorus loading to Lake Sallie may be assimilated to a great extent in macrophyte growth, rather than by algal uptake. This would keep both the algal and mean phosphorus concentrations in Lake Sallie lower than expected from its reported phosphorus loading. This would explain why Figure 15, based on watershed land usage, indicates a possible phosphorus loading underestimation for Lake Sallie while Figure 14, based partly on mean phosphorus concentration, indicates the phosphorus loading to be reasonable. Any estimation of trophic state, based on Lake Sallie's algal characteristics alone, would likely indicate a trophic condition which is consistent with that indicated by Neel (1975), but which is not a realistic appraisal of the overall degree of the fertility of Lake Sallie because it ignores the portion of Lake Sallie's primary productivity which is manifested in macrophyte growth.

SECTION VIII

US OECD EUTROPHICATION STUDY NITROGEN DATA:

AS APPLIED IN VOLLENWEIDER NITROGEN LOADING AND MEAN DEPTH/HYDRAULIC RESIDENCE TIME RELATIONSHIP

In addition to phosphorus loadings, the Vollenweider relationship can also be applied to total nitrogen loadings. However, because of the relatively scant knowledge concerning nitrogen relationships in natural waters, Vollenweider has not developed the permissible and excessive boundary conditions for a nitrogen loading-mean depth/hydraulic residence time relationship. Thus, the trophic state of a water body which is nitrogen limited with respect to aquatic plant nutrient requirements cannot be determined in the same manner as with Vollenweider's phosphorus loading diagram. Conceptually, such an application is possible. However, it would necessarily be more difficult to establish the permissible and excessive nitrogen loading boundary lines on such a loading diagram.

As indicated earlier, several approaches could be utilized to develop critical nitrogen loadings for lakes. One of the most obvious involves using a direct proportion between the critical N and P loadings based on typical algal stoichiometry of 16 nitrogen atoms for every phosphorus atom. On a mass basis, this would mean that the permissible nitrogen loadings would be increased by approximately 7.5 times the corresponding phosphorus loadings.

Another approach would be utilization of the equivalent nitrogen concentrations developed by Sawyer (1947). The validity for this approach stems from the fact that Sawyer's critical phosphorus concentrations play a dominant role in establishing the permissible and excessive lines on the Vollenweider phosphorus loading relationship. Sawyer suggested a critical inorganic nitrogen concentration of 0.3 mg N/l. There are a number of potential problems involved in attempting to use a direct proportion between nitrogen and phosphorus critical loads, the most important of which would occur in highly eutrophic lakes, where nitrogen, rather than phosphorus, is frequently the key limiting element. In these water bodies, blue-green algae, some of which are nitrogen fixers, often dominate. While nitrogen fixation does occur in many lakes, its overall significance is poorly understood. It does not appear, as sometimes stated, that nitrogen fixation prevents lakes from becoming nitrogen limited. There are some lakes which show significant nitrogen limitation in the presence of nitrogen-fixing algae. Torrey and Lee (1976), studying Lake Mendota, found that less than 10 percent of the total nitrogen input was from nitrogen fixation.

Eutrophic lakes frequently show appreciable denitrification reactions in which nitrate is converted to nitrogen gas in anoxic waters and sediments. This type of reaction would tend to convert readily available nitrogen into unavailable forms. Brezonik and Lee (1968) determined the significance of denitrification as a means of removing nitrogen from Lake Mendota.

Probably one of the most significant problems with trying to develop a similar set of relationships for nitrogen as have been presented for phosphorus is that it is often more difficult to accurately estimate nitrogen loads. Potentially significant problems occur with estimations of nitrogen input from groundwater, which can be an appreciable nitrogen source for some lakes. As discussed by Sonzogni and Lee (1974), even if the groundwater input and its associated nitrate content are known, one cannot be certain of the degree of nitrification, if any, that will occur when the groundwater nitrate comes in contact with the lake sediments.

The total nitrogen loading diagram, containing the data for the US OECD water bodies, is presented in Figure 21. The data was presented in Table 20. The total nitrogen loadings is comprised of the inorganic nitrogen fraction (i.e., NO_3^- + NO_2^- + NH_4^+ as N), plus the organic nitrogen fraction, except as indicated. There are fewer data points in Figure 21 than in Figure 19 because nitrogen loadings were not reported for all the US OECD water bodies.

If one compares the nitrogen loading diagram (Figure 21) with the phosphorus loading diagram (Figure 19), an interesting observation is that, except for the order of magnitude difference on the loading axis, there is a good agreement between the relative positions of the common water bodies on both the loading diagrams. The relative zones denoting the different trophic states on the phosphorus loading diagram are also maintained on the nitrogen loading diagram. This similarity implies that a water body receives nutrients in a relatively constant ratio, with the nitrogen loading being approximately one order of magnitude greater than the phosphorus loading. This is consistent with the view that different types of land usage within a watershed will yield a relatively constant amount of nutrient export over the annual cycle. In addition, the ratio of nitrogen to phosphorus of ten to one is approximately at the boundary condition between limiting nutrients (i.e., above an N:P mass ratio of about eight to one, phosphorus is the limiting aquatic plant nutrient; below an eight to one ratio, nitrogen appears to be the limiting

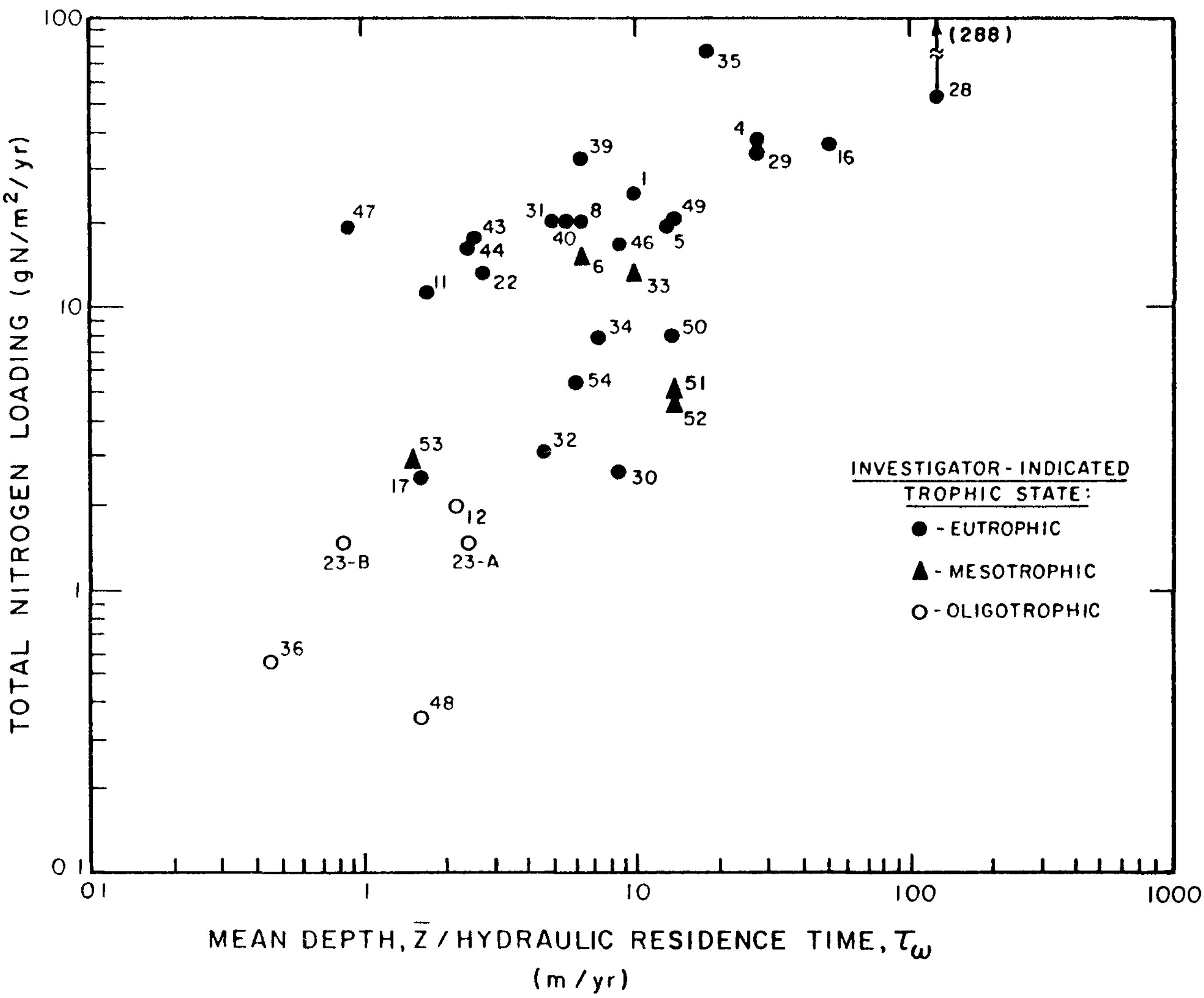

Figure 21. US OECD Data Applied to Vollenweider Nitrogen Loading and Mean Depth/Hydraulic Residence Time Relationship

nutrient -- see Tables 9 and 10). This implies nitrogen and phosphorus are present in such constant relative amounts that either nutrient could become limiting with a small relative increase in the other. Such a view is consistent with a water body being phosphorus-limited during one time of the year and nitrogen-limited during another time of the year (i.e., Lake Mendota). It is also consistent with nitrogen limitation in one portion of a water body and phosphorus limitation in another portion of the same water body at the same time because of different land usage patterns in different portions of the watershed (i.e., Potomac Estuary -- see Table 9).

There is no equivalent expression for Vollenweider's mean phosphorus/influent phosphorus concentration relationship (Equation 26) to check the US OECD nitrogen loading estimates. There is also no equivalent expression for Vollenweider's critical phosphorus loading relationship (Equation 19) which can be applied to the US OECD water body loadings. However, it is possible to compare the reported nitrogen loadings with those predicted with the watershed land use nitrogen export coefficient calculations. This was done earlier for the US OECD water bodies (Figure 16). The US OECD data were presented in Table 18. The nitrogen watershed land use export and atmospheric input coefficients used by these reviewers were taken from Table 17.

Figure 16 indicates generally good agreement between the predicted and reported nitrogen loadings for the US OECD water bodies. As with the phosphorus loadings, a nitrogen loading was considered reasonable if it was within two-fold above or below the nitrogen loading predicted with the use of the watershed land use nitrogen export calculations. However, it should be noted that most US OECD investigators did not report data for dry fallout and nitrogen fixation in their nitrogen inputs (Table 13). If the results of Figure 16 are correct, this suggests these sources are not significant nitrogen inputs to the US OECD water bodies when they are compared to the other nitrogen inputs. This is inconsistent with the observations of Kluesener (1972) and Sonzogni and Lee (1974) who reported that nitrogen inputs from these two sources could be substantial.

COMPARISON OF RESULTS:

Discrepancies Between Investigator-Indicated Nitrogen Loadings and Watershed Nitrogen Export Coefficient Calculations

There are a few US OECD water bodies in Figure 16 whose reported nitrogen loadings are indicated as possibly being in error. These include Lake Sallie (32), Lake Sammamish (33), Lake Tahoe (36), East Twin Lake-1972 (39), West Twin Lake-1972 (43), and Lake Waldo (48). Among the US OECD water bodies whose nitrogen loadings are indicated in Figure 16 as possibly being in error, only Lakes Sallie (32), Tahoe (36) and Waldo (48) may be nitrogen-limited.

Lake Sallie --

Lake Sallie is indicated as having a nitrogen loading underestimation of approximately thirty-fold. In Vollenweider's phosphorus loading diagram (Figure 19), Lake Sallie plots in a zone indicative of a relatively advanced eutrophic condition. However, as Lake Sallie may not be phosphorus-limited (Table 9), this predicted trophic condition in Figure 19 may not be indicative of Lake Sallie's true trophic state. In fact, Neel (1975) has characterized Lake Sallie as being in a late mesotrophic-early eutrophic condition. Neel (1975) has also indicated that phosphorus does not appear to control algal growth in Lake Sallie beyond a certain point. This is consistent with observations made by Vollenweider (1975a) that as a water body becomes more eutrophic, beyond a certain point nitrogen becomes the limiting nutrient, even though phosphorus may initially have been limiting aquatic plant growth. According to Vollenweider, the turning point is reached when the ratio of the nitrogen residence time to the phosphorus residence time drops below a value of one. However, only the inorganic nitrogen concentration for Lake Sallie was reported. Calculation of the nitrogen residence time requires the total (i.e., organic fraction + inorganic fraction) nitrogen concentration be known. Therefore, calculation of the ratio of the residence times of nitrogen to phosphorus is not possible for Lake Sallie (see Table 21). As a result, it is not clear whether nitrogen or phosphorus limits algal growth in Lake Sallie.

Lake Tahoe --

The nitrogen loading estimate for Lake Tahoe (36) is indicated in Figure 16 as being underestimated about four-fold. This water body is classified as ultra-oligotrophic by Goldman (1975) and by its position on the Vollenweider phosphorus loading diagram (Figure 19). It also plots in the lower half of the nitrogen loading diagram (Figure 21), implying an oligotrophic status. Lake Tahoe is nitrogen-limited (Table 9) according to its investigator.

The atmospheric nitrogen contributions for Lake Tahoe were considered insignificant by Goldman (1975). However, several investigators (Kluesener, 1972; Sonzogni and Lee, 1974; Murphy, 1974) have indicated this can be a significant nutrient source, especially for oligotrophic water bodies. In addition, the nitrogen contribution from nitrogen fixation was not considered in the nitrogen loading estimate for Lake Tahoe, though this latter source is likely small.

The present condition of Lake Tahoe indicates it to be much closer to its limit of permissible nutrient loading than originally thought (Vollenweider and Dillon, 1974). Thus, the nitrogen loadings to Lake Tahoe may have been underestimated to some degree. However, it is not clear that the reported nitrogen loadings have been underestimated by the factor of four indicated in Figure 16.

Lake Sammamish, Lake Cayuga and Twin Lakes --

Lake Sammamish, East Twin Lake-1972 and West Twin Lake-1972 show apparent nitrogen loading overestimations based on Figure 16. Dry fallout and nitrogen fixation contributions were not considered in the nitrogen loading estimates for these water bodies. As a result, one would expect the nitrogen loadings to be underestimated, rather than overestimated, unless the nitrogen loadings from one or more of the sources have been highly overestimated. The possible nitrogen loading overestimations of approximately two-fold for the Twin Lakes (East Twin Lake-1972 (39) and 1974 (40) and West Twin Lake-1972 (43) and 1973 (44)) indicated in Figure 16 are likely in error. The nitrogen loading for Cayuga (6) is also possibly overestimated by nearly two-fold. The nitrogen loadings reported for these three water bodies comprise only the inorganic nitrogen fractions of the total nitrogen loading. They do not include the organic nitrogen fraction. While the organic nitrogen fraction is not immediately available for algal growth, Cowen et al. (1976a; 1976b) have reported that, under optimal conditions, 50 to 80 percent of the organic nitrogen fraction present in urban and rural runoff can be converted, in a few weeks to several months, to inorganic nitrogen forms available for algal growth. Consequently, omission of the organic nitrogen fraction can result in a gross underestimation of the total nitrogen loading to a water body in an urban or rural area. It would seem that these three water bodies could not exhibit the nitrogen loading overestimation indicated in Figure 16 unless the inorganic nitrogen fraction of the total nitrogen loading has been grossly overestimated. As a result, the overestimation of the nitrogen loadings indicated in Figure 16 for the Twin Lakes and Lake Cayuga may be in error.

In general, the nitrogen loadings for most of the US OECD water bodies, when compared with the nitrogen loadings derived from watershed land use nitrogen export coefficients, appear to be reasonable. This supports the view of these reviewers that the use of a nitrogen loading diagram for denoting trophic state associations for nitrogen-limited water bodies, similar to the Vollenweider phosphorus loading diagram for phosphorus-limited water bodies (i.e., Figure 19), is plausible. Such an application, however, must wait until a valid input-output model similar to that derived for phosphorus (Vollenweider, 1975a) is available for nitrogen loadings.

SECTION IX

US OECD DATA APPLIED IN OTHER NUTRIENT RELATIONSHIPS

US OECD PHOSPHORUS DATA APPLIED IN VOLLENWEIDER'S PHOSPHORUS LOADING CHARACTERISTICS AND MEAN CHLOROPHYLL RELATIONSHIP

As indicated earlier, several investigators have demonstrated a relationship between phosphorus concentration at spring overturn and the annual or summer chlorophyll concentrations (Sawyer, 1947; Sakamoto, 1966; Dillon, 1974a; Dillon and Rigler, 1974a; Jones and Bachmann, 1976). A positive correlation between these parameters was also illustrated by Vollenweider at the May, 1975 North American OECD meeting in Minneapolis. Consequently, Vollenweider (1976a) developed a diagram for predicting algal biomass, expressed as chlorophyll concentration, as a function of a water body's specific phosphorus loading characteristics. The derivation of this approach was presented in an earlier section of this report (see Equation 20 and Figure 11). The reader is reminded that this phosphorus loading expression $(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$ is equivalent to the predicted in-lake steady state mean phosphorus concentration. In Equation 20 (used in Figure 22), the phosphorus loadings can be checked as a function of the term $L(P)/q_s$ and related to the mean in-lake phosphorus concentration. A similar approach was used to check the phosphorus loading estimates, as illustrated in Figure 14 and Equations 25 and 26.

The phosphorus loading characteristics and epilimnetic mean chlorophyll *a* diagram is presented in Figure 22 for the US OECD water bodies. The pertinent data for this diagram are presented in Table 23. If a data range was reported for a water body, the mean value was used in all calculations.

Based on Sawyer's (1947) and Sakamoto's (1966) critical nutrient concentrations, oligotrophic water bodies will plot to the left of the 10 mg/m^3 phosphorus loading characteristics level, and eutrophic water bodies to the right of the 20 mg/m^3 phosphorus loading characteristics level. The mesotrophic water bodies would plot between these two loading levels. The relative degree of eutrophy or oligotrophy of a water body is determined by its horizontal displacement to the right or left of the 10 mg/m^3 phosphorus loading characteristics level (i.e., predicted in-lake steady state phosphorus concentration). Thus, this 10 mg/m^3 concentration corresponds to Vollenweider's (Figure 19) permissible phosphorus loading.

Table 23. US OECD DATA APPLIED TO VOLLENWEIDER'S PHOSPHORUS LOADING AND MEAN CHLOROPHYLL *a* CONCENTRATION RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading, $L(P)$ $(mg/m^2/yr)$[b]	Mean Depth, $\bar{z}$ (m)	Hydraulic Loading, q_s (m/yr)[c]	$\frac{L(P)/q_s}{1+\sqrt{\bar{z}/q_s}}$	Mean Secchi Depth (m)	Mean Chlorophyll *a* Concentration (µg/l)[b]
Blackhawk (1)[d]	E	2130-2320	4.9	9.8	133	3.6	15[e]
Brownie (2)	E	1180	6.8	3.4	144	1.5	6[f]
Calhoun (3)	E	860	10.6	2.9	101	2.1	6[f]
Camelot-Sherwood Complex (4)	E	2350-2680	3	21.4-33.3	69	2.0	6[3]
Canadarago 1968 (5-A)	E	800	7.7	12.8	35.1	-	13
1969 (5-B)	E	800	7.7	12.8	35.1	1.8	7
Cayuga 1972 (6-A)	M	800	54	6.3	32.4	2.3	6
1973 (6-B)	M	800	54	6.3	32.4	2.3	5
Cedar (7)	E	350	6.1	1.8	69.0	1.8	20[f]
Cox Hollow (8)	E	1620-2080	3.8	5.4-7.6	160	1.5	26[e]
Dogfish 1972 (10)	O	20	4	1.1	6.3	2.5	4 (2)[g]
Dutch Hollow (11)	E	950-1010	3	1.7	246	0.8	34[e]
George (12)	O-M	70	18	2.2	8.3	8.5	-
Harriet (13)	E	710	8.8	3.7	75	2.4	4[f]

Table 23 (continued). US OECD DATA APPLIED TO VOLLENWEIDER'S PHOSPHORUS LOADING AND MEAN CHLOROPHYLL *a* CONCENTRATION RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Loading, L(P) ($mg/m^2/yr$)[b]	Mean Depth, $\bar{z}$ (m)	Hydraulic Loading, q_s (m/yr)[c]	$\frac{L(P)/q_s}{1 + \sqrt{\bar{z}/q_s}}$	Mean Secchi Depth (m)	Mean Chlorophyll *a* Concentration (µg/l)[b]
Isles (14)	E	2030	2.7	4.5	254	1.0	53[f]
Kerr Reservoir	E-M						
Roanoke Arm (16)	-	5200	10.3	51.5	69.8	1.4	13
Nutbush Arm (17)	-	700	8.2	1.6	134	1.2	21
Lamb							
1972 (19)	O	30	4	1.7	7.0	2.2	3 (3)[g]
Meander							
1972 (21)	O	30	5	1.8	6.3	3.0	2 (1)[g]
Mendota (22)	E	1200	12	2.7	142	3.0	10 (20)[h]
Michigan (Open Waters) (23-A)	O	140	84	2.8	7.7	-	2
Lower Lake (23-B)	O	140	84	0.84	15.2	-	2
Minnetonka							
1969 (25)	E	500	8.3	1.3	109	1.5	21
1973 (26)	E→M	100 (180)[k]	8.3	1.3	21.9 (39.4)[k]	1.8	12
Potomac Estuary	U-E						
Upper (28)	-	85000	4.8	120	590	0.6	30-150
Middle (29)	-	8000	5.1	28.3	198	0.9	30-100
Lower (30)	-	1200	7.2	8.5	73.4	1.6	10-20
Redstone (31)	E	1440-1680	4.3	4.3-6.1	156	1.6	13[e]

Table 23 (continued). US OECD DATA APPLIED TO VOLLENWEIDER'S PHOSPHORUS LOADING AND MEAN CHLOROPHYLL a CONCENTRATION RELATIONSHIP

Water Body		Trophic State[a]	Phosphorus Loading, L(P) ($mg/m^2/yr$)[b]	Mean Depth, $\bar{z}$ (m)	Hydraulic Loading, q_s (m/yr)[c]	$\frac{L(P)/q_s}{1+\sqrt{\bar{z}/q_s}}$	Mean Secchi Depth (m)	Mean Chlorophyll a Concentration (µg/l)[b]
Sallie	(32)	E	1500-4200	6.4	3.6-5.8	275	-	-
Sammamish	(33)	M	700	18	10	29.9	3.3	5
Shagawa	(34)	E	700	5.7	7.1	52.0	2.3	15 (24)[i]
Stewart	(35)	E	4820-8050	1.9	23.8	211	1.4	12[e]
Tahoe	(36)	U-O	50	313	0.45	4.0	28.3	< 1[g]
East Twin								
1972	(39)	E	700(700)[k]	5	6.2	59.6	1.6	26
1973	(40)	E	500(500)	5	5.6	45.8	2.3	22
1974	(41)	E	700(500)	5	10	41.0	1.9	28
West Twin								
1972	(43)	E	400(400)[k]	4.3	2.7	65.4	2.2	40
1973	(44)	E	300(200)	4.3	2.4	53.4	2.8	23
1974	(45)	E	300(300)	4.3	4.3	34.9	2.3	28
Twin Valley	(46)	E	1740-2050	3.8	7.6-9.5	133	1.5	19[e]
Virginia	(47)	E	1150-1480	1.7	0.6-1.9	44.6	1.2	29[e]
Waldo	(48)	U-O	17	36	1.7	1.8	28.0	< 1[j]

Table 23 (continued). US OECD DATA APPLIED TO VOLLENWEIDER'S PHOSPHORUS LOADING AND MEAN CHLOROPHYLL a CONCENTRATION RELATIONSHIP

Water Body		Trophic State[a]	Phosphorus Loading, L(P) ($mg/m^2/yr$)[b]	Mean Depth, $\bar{z}$ (m)	Hydraulic Loading, q_s (m/yr)[c]	$\frac{L(P)/q_s}{1 + \sqrt{\bar{z}/q_s}}$	Mean Secchi Depth (m)	Mean Chlorophyll a Concentration ($\mu g/l$)[b]
Washington								
1957	(49)	E	1200	33	13.8	34.1	2.2	12
1964	(50)	E	2300	33	13.8	65.3	1.2	20
1971	(51)	M	430	33	13.8	12.2	3.5	6
1974	(52)	M	470	33	13.8	13.4	3.8	- (4)
Weir	(53)	M	140	6.3	1.5	30.6	1.9	8
Wingra	(54)	E	900	2.4	6	91.9	1.3	-

EXPLANATION:

[a]Investigator-indicated trophic states: E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

[b]Based on investigator's estimates.

[c]Hydraulic loading, $q_s = \bar{z}/t_\omega$ = hydraulic residence time = water body volume (m^3)/annual inflow volume (m^3/yr).

[d]( ) = Identification number for Figures 22, 23 and 24 (see Table 14)

Table 23 (continued). US OECD DATA APPLIED TO VOLLENWEIDER'S PHOSPHORUS LOADING AND MEAN CHLOROPHYLL _a_ CONCENTRATION RELATIONSHIP

EXPLANATION (continued)

[e] First two meters of water column.

[f] Summer surface values.

[g] Euphotic zone.

[h] Growing season.

[i] Ice-free period.

[j] Average value for August.

[k] All data in parentheses represent data received by these reviewers from the principal investigators after the completion of this report. Figure 22 is based on the original data reported by the investigators and does not reflect these revised values. Examination of the revised data indicated no significant changes in the overall conclusions concerning these water bodies.

Dash (-) indicates data not available.

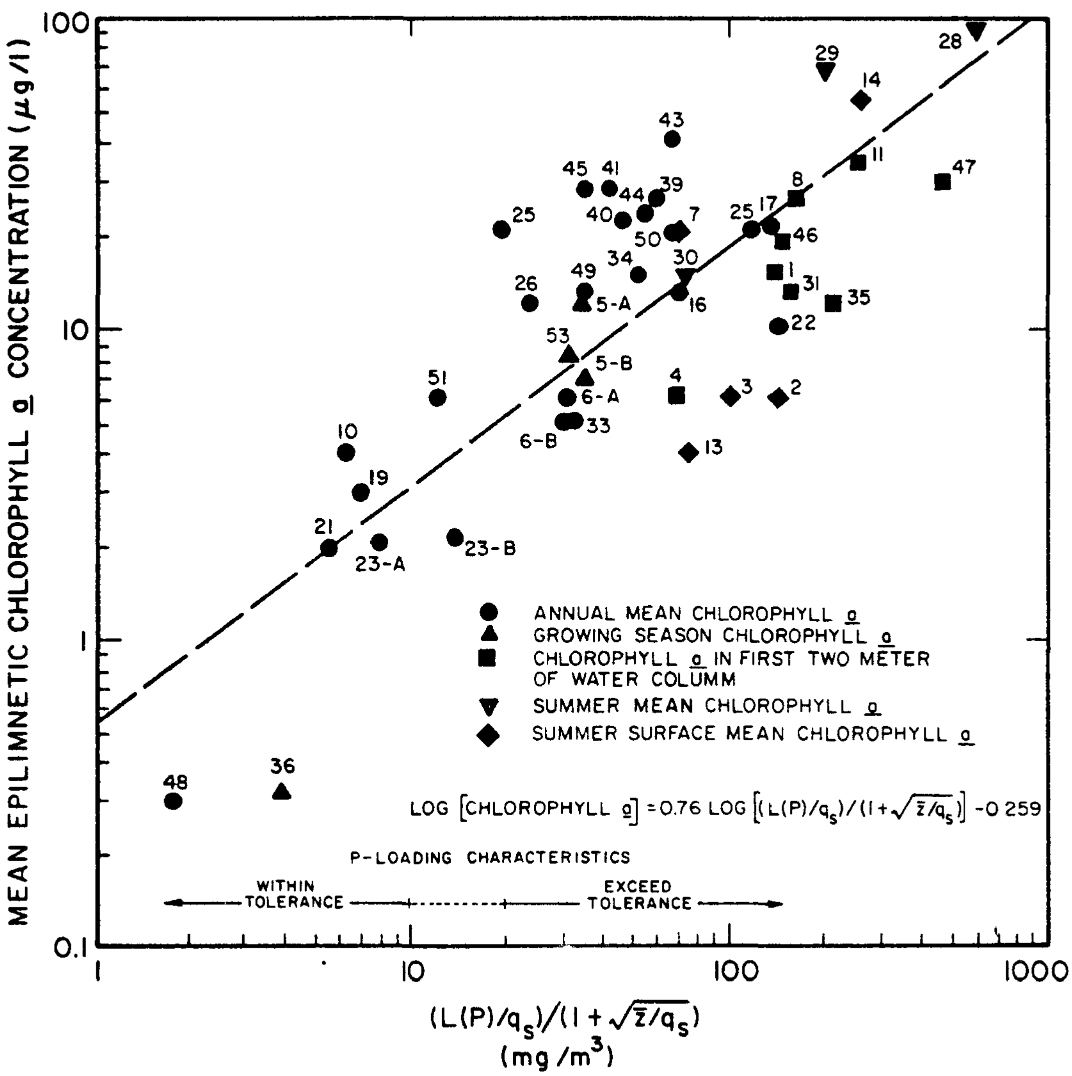

Figure 22. US OECD Data Applied to Vollenweider Phosphorus Loading Characteristics and Mean Chlorophyll $\underline{a}$ Relationship

Examination of Figure 22 indicates the investigators have used a variety of approaches for estimating the chlorophyll a content of their water. Some reported values are summer means while other values are annual means. Some values are means for the euphotic zone while others are means for the first two meters of the water column. Therefore, in a strict sense the reported chlorophyll a data for the US OECD water bodies are not directly comparable. However, even with these limitations, there is reasonable agreement ($r = 0.77$) between the predicted trophic states of the US OECD water bodies, based on their position to the right or left of the 10 mg/m^3 permissible phosphorus concentration boundary line and the investigator's subjective trophic state characterizations. In general, the results of Figure 22 confirm the results indicated in the Vollenweider phosphorus loading diagram (Figure 19).

Figure 22 also supports some of the possible phosphorus loading estimate discrepancies indicated in Figures 14 and 15. For example, based on its phosphorus loading characteristics and mean chlorophyll a concentrations, Lake Weir plots in the eutrophic zone in Figure 22, in disagreement with the mesotrophic condition indicated by Brezonik and Messer (1975). This supports the possibility that the phosphorus loading underestimation indicated in Figures 14 and 15 are in error. If, on the other hand, the phosphorus loading estimates for Lake Weir are correct, then the level of chlorophyll production per unit of input phosphorus must be higher in Lake Weir than in other water bodies. This would support the idea of a different phosphorus loading-algal response relationship in warm water bodies compared to that found in water bodies in the north temperate zones of the US. Furthermore, the relative closeness of Lake Dogfish (10), Lake Lamb (19) and Lake Meander (31) to the 10 mg/m^3 concentration mark in Figure 22 supports the possible phosphorus loading underestimations indicated earlier in Figure 15 for these water bodies. As indicated earlier, their reported phosphorus loadings place them in the trophic zone of the Vollenweider phosphorus loading diagram (Figure 19) characteristic of ultra-oligotrophic Lakes Tahoe and Waldo. However, Lakes Dogfish, Lamb and Meander are clearly more productive, in terms of relative chlorophyll a concentrations, than Lakes Tahoe and Waldo, supporting the phosphorus loading underestimation indicated in Figure 15 for these three water bodies.

In spite of the non-uniform computations of the mean chlorophyll a concentrations used in Figure 22, the results of this relationship between phosphorus loading characteristics, (i.e., predicted in-lake phosphorus concentration - see Equation 20) and chlorophyll a concentrations indirectly support the validity of the Vollenweider phosphorus loading diagram criteria illustrated in Figure 19.

US OECD PHOSPHORUS DATA APPLIED IN PHOSPHORUS LOADING AND SECCHI DEPTH RELATIONSHIP

The use of the Secchi depth as an indicator of algal biomass has recently been proposed by several investigators (Edmondson, 1972; Carlson, 1974; Shapiro, 1975b; Shapiro et al., 1975). The use of this parameter as an indicator of a water body's trophic condition is based largely on the public's perception of eutrophication problems. Remedial treatment programs, including sewage diversion and advanced waste treatment, have often been initiated because of the public's reaction to the side effects of eutrophication, such as dense algal blooms or decaying algal mats. As a result, water transparency or clarity has probably become the most frequently cited all-around general indicator of water quality. The higher the transparency of the water body, the higher is thought to be the general water quality. Obvious exceptions to this general rule would be water bodies with high color content.

Edmondson (1972) has found a close relationship between Secchi depth and algal biomass (expressed as chlorophyll concentration) in Lake Washington. While there are likely some effects due to light scattering by non-planktonic particles in the water, there is a definite negative hyperbolic relationship between Secchi depth and chlorophyll concentration, with the slope of the curve steepest at the lower biomass levels. This indicates changes in biomass, as reflected in chlorophyll concentrations, are more easily detected in clear (i.e., oligotrophic) waters than in eutrophic waters. Above approximately 20 μg/l chlorophyll concentrations, at least in Lake Washington, a large increase in mean chlorophyll does not produce a proportionately large decrease in Secchi depth. This indicates that, above a certain degree of eutrophication, Secchi depth readings lose sensitivity as an indicator of changes in algal biomass, other than a low Secchi depth indicating a relatively eutrophic condition of the water body.

Even with this limitation, however, the use of Secchi depth measurements as an indicator of a water body's algal biomass, and hence general trophic condition, remains an easily measured parameter, involving a minimum of time and cost. In addition, its meaning is easily understood by the general public and is a parameter which can be evaluated over time in correlation with the general trophic condition of the water body.

As the algal biomass of a water body is related to its nutrient flux, the Secchi depths of the US OECD water bodies were examined as a function of their phosphorus loading characteristics in a manner analagous to that of chlorophyll a concentration in Figure 22. In order to give the plot the same general slope as expressed in Vollenweider's chlorophyll concentration versus phosphorus loading characteristics, the reciprocal of the Secchi depth was plotted versus the phosphorus loading expression, $(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$. The pertinent data was presented in Table 23. The US OECD eutrophication study data are presented in Figure 23.

Examination of Figure 23 shows a definite relationship does exist between Secchi depth and phosphorus loadings, with the reciprocal of the Secchi depth increasing as a function of the phosphorus loading. However, the slope is not as steep as that indicated in Figure 22 between chlorophyll a concentration and phosphorus loading characteristics. Particularly scattered are the data sets for the oligotrophic and mesotrophic water bodies.

In an attempt to graphically produce a greater spread of data, a semilog plot of the US OECD data was prepared. This is illustrated in Figure 24. Examination of Figure 24 again shows a relationship exists between these two parameters. As the phosphorus loading increases, the reciprocal of the Secchi depth also increases, with the steepest slope at the higher phosphorus loading and lower Secchi depth values. However, the data sets still exhibit considerable scatter. Unfortunately, there is not a sufficient number of oligotrophic water bodies in the US OECD eutrophication study to allow examination of this relationship, using US OECD data, other than on a general qualitative basis. As a nonlinear relationship exists between Secchi depth and chlorophyll (Edmondson, 1972), it is not surprising to see a nonlinear relationship existing between phosphorus loading and Secchi depth, particularly since the algal biomass in a water body is generally a function of the intensity of the nutrient flux. The use of this relationship as a tool for assessing the expected change in water quality resulting from a changed phosphorus load will be discussed in a later section of this report.

US OECD PHOSPHORUS DATA APPLIED IN DILLON'S PHOSPHORUS LOADING-PHOSPHORUS RETENTION AND MEAN DEPTH RELATIONSHIP

A different type of phosphorus loading diagram was subsequently developed by Dillon (1975; Vollenweider and Dillon, 1974). This loading diagram considers not only the phosphorus loading to a water body, but also the capacity of the water body to retain the input phosphorus. Vollenweider's earlier relationships do this implicitly as a function of mean depth, $\bar{z}$, or hydraulic loading, q_s. Derivation of Dillon's model was presented in an earlier section of this report. Dillon's relationship allows one to consider the effects of flushing time, phosphorus loading and phosphorus retention on the degree of fertility of a water

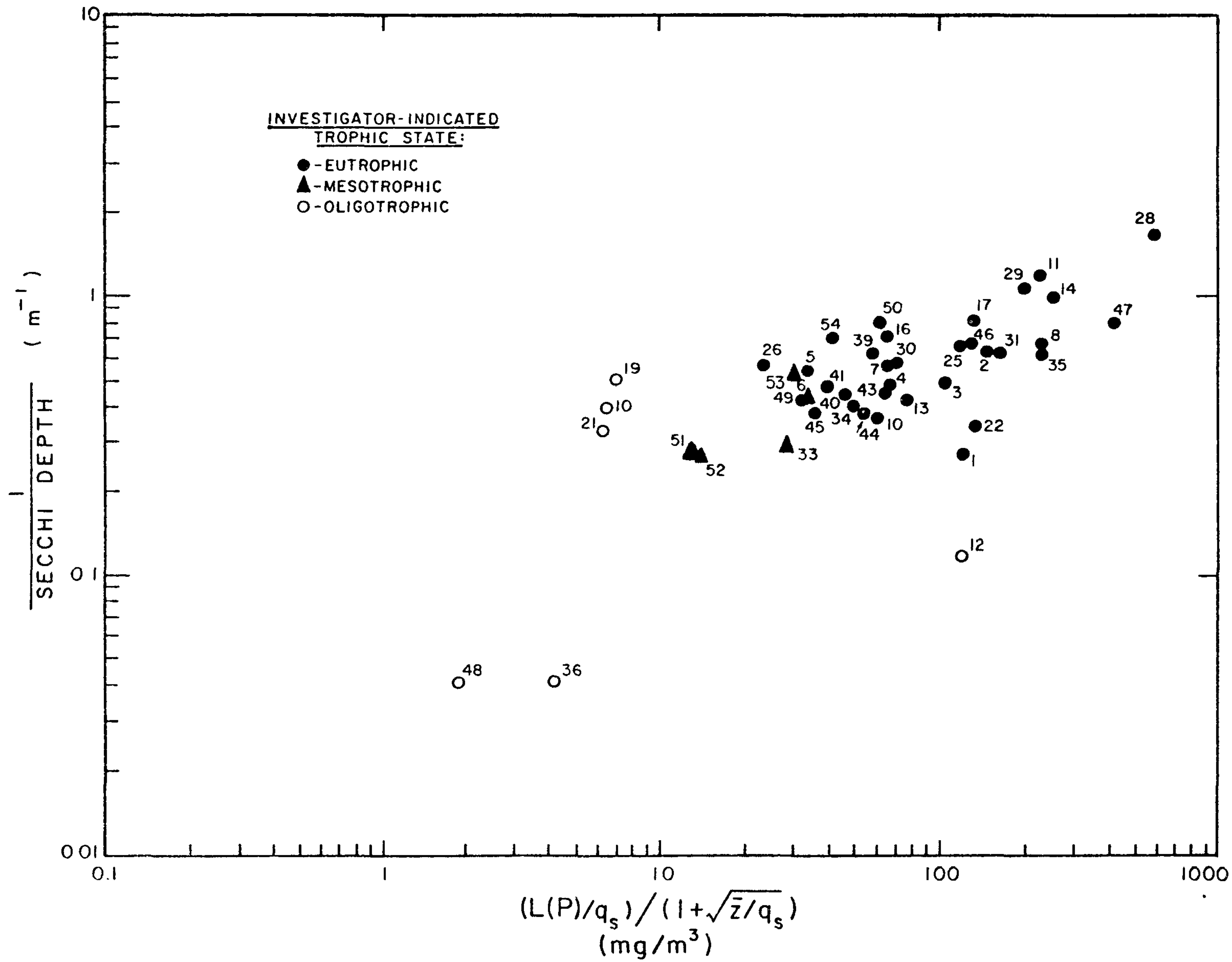

Figure 23. US OECD Data Applied to Phosphorus Loading and Secchi Depth Relationship (Log-Log Scale).

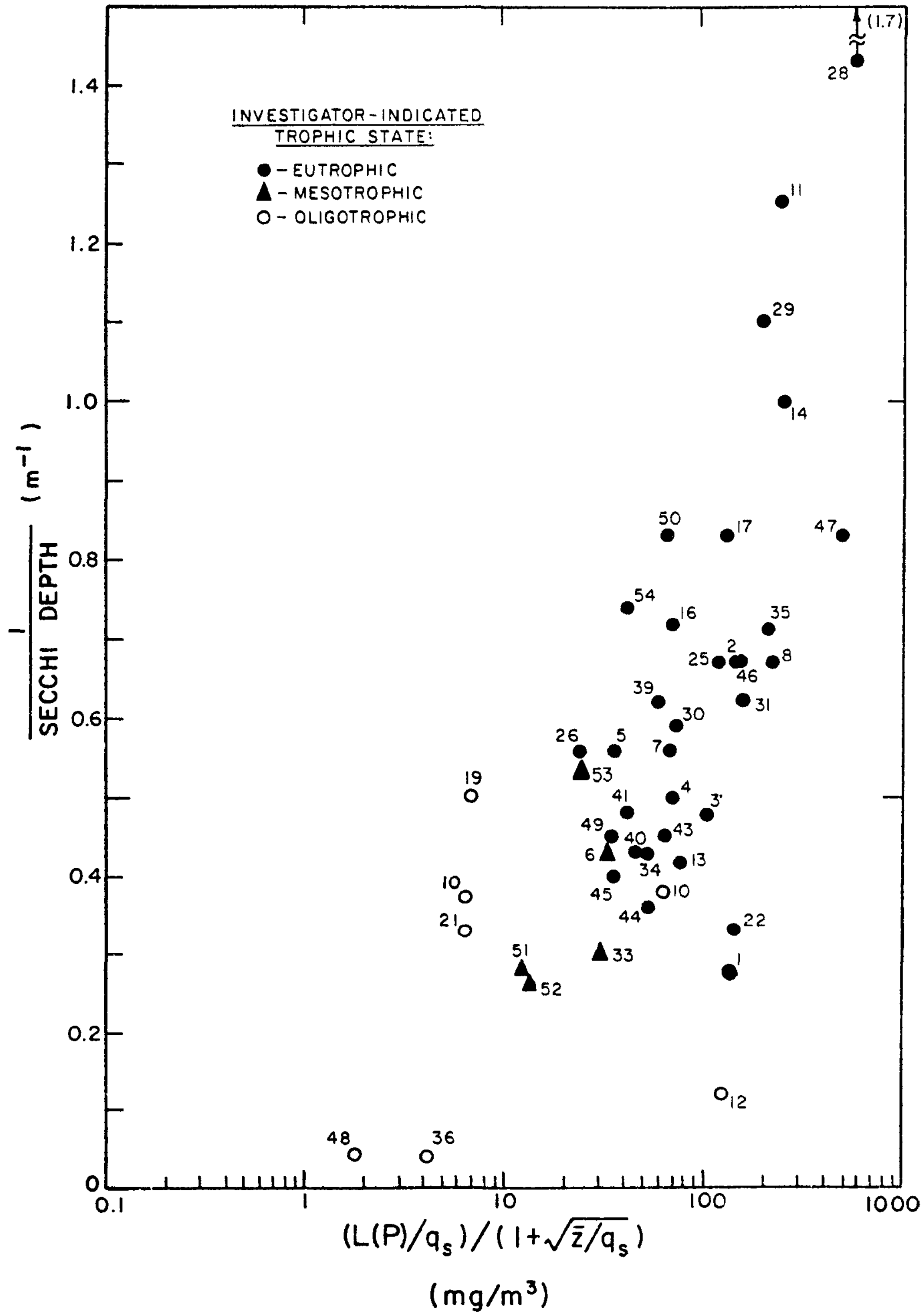

Figure 24. US OECD Data Applied to Phosphorus Loading and Secchi Depth Relationship (Semilog Scale).

body. A main feature of Dillon's model is that since a water body's phosphorus retention capacity is a function of its flushing rate, consideration of the phosphorus retention coefficient allows for a more accurate determination of the effects of an extremely fast or slow hydraulic flushing rate on the phosphorus loading-trophic response relationship.

Dillon's phosphorus loading diagram is presented in Figure 25. The pertinent US OECD data are presented in Table 24. If a data range was reported for a water body, the mean value was used in all calculations. Phosphorus concentration boundary conditions of 10 μg/l and 20 μg/l (Sawyer, 1947; Sakamoto, 1966; Dillon, 1975) correspond to Vollenweider's permissible and excessive loading lines, respectively (Figure 19). The trophic state associations are similar to those in Figure 19.

As was found with Vollenweider's phosphorus loading diagram (Figure 19), water bodies of similar trophic character plot in the same relative area in Dillon's loading diagram (Figure 25). There is generally good agreement between the predicted trophic states in Dillon's loading diagram and the US OECD investigator-indicated trophic states. In addition, Figure 25 supports the possibility of a phosphorus loading underestimation for Lakes Dogfish (10), Lamb (19), and Meander (21), indicated in earlier diagrams.

In general, Dillon's phosphorus loading diagram appears to be a valid procedure for establishing the relative trophic conditions and phosphorus concentrations of water bodies. It also indirectly supports the validity of the Vollenweider phosphorus loading relationship expressed in Figure 19. It should be mentioned, however, that while Dillon's phosphorus loading diagram is a substantial improvement over Vollenweider's original phosphorus loading and mean depth diagram (Figure 5), it does not appear to offer any significant improvement over the information obtained with Vollenweider's modified phosphorus loading and mean depth/hydraulic residence time loading diagram (Figure 19). Rather, it is an alternate method for predicting the relative degree of fertility of a water body. In fact, Dillon (Vollenweider and Dillon, 1974) offers his model as a simple method for predicting phosphorus concentrations rather than as a substitute for Vollenweider's modified phosphorus loading diagram. It should be mentioned that Vollenweider's relationship used in Figure 19 (i.e., Equation 9) assumes that R(P) is expressed solely through the hydraulic residence time, τ_ω. However, Vollenweider's relationship likely would not indicate if any other parameters affected R(P). In this regard, Dillon's relationship may be more complete.

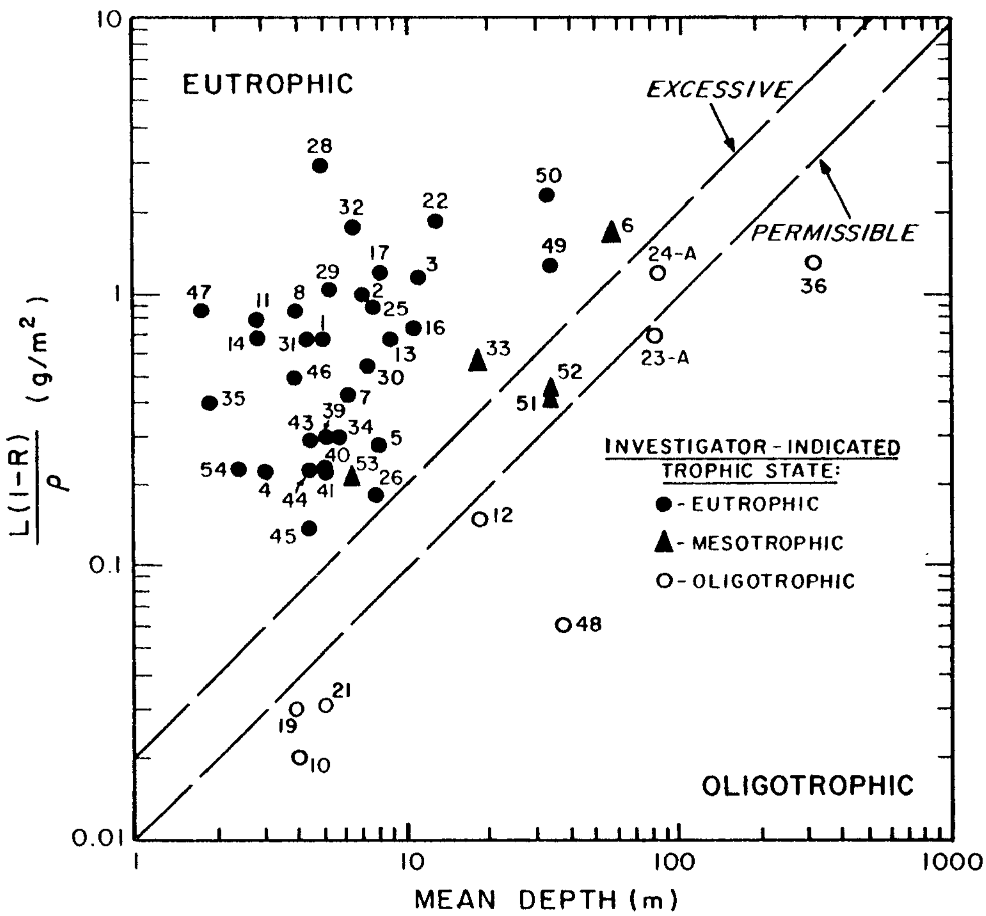

Figure 25. US OECD Data Applied to Dillon Phosphorus Loading - Phosphorus Retention and Mean Depth Relationship

Table 24. US OECD DATA APPLIED TO DILLON'S PHOSPHORUS LOADING-PHOSPHORUS RETENTION AND MEAN DEPTH RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Retention Coefficient, R[b]	Phosphorus Loading, L ($g/m^2/yr$)[c]	Flushing Rate, ρ (yr^{-1})[d]	$L(1-R)/\rho$ (mg/m^2)	Mean Depth, $\bar{z}$ (m)
Blackhawk (1)[e]	E	0.41	2.1-2.3	2.0	0.70	4.9
Brownie (2)	E	0.59	1.18	0.5	0.98	6.8
Calhoun (3)	E	0.66	0.86	0.28	1.05	10.6
Camelot-Sherwood(4)	E	0.23-0.27	2.35-2.68	7.1-11.1	0.21	3.0
Canadarago	E	0.44	0.8	1.67	0.27	7.7
Cayuga (6)	M	0.75	0.8	0.12		54
Cedar (7)	E	0.64	0.35	0.30	0.41	6.1
Cox Hollow (8)	E	0.41-0.46	1.62-2.08	1.4-2.0	0.61	3.8
Dogfish (10)	O	0.65	0.02	0.29	0.02	4.0
Dutch Hollow (11)	E	0.57	0.95-1.01	0.56	0.75	3.0
George (12)	O-M	0.74	0.07	0.12	0.15	18
Harriet (13)	E	0.61	0.71	0.42	0.66	8.8
Isles (14)	E	0.44	2.03	1.67	0.69	2.7
Kerr Reservoir	E-M					
Roanoke (16)	-	0.31	5.2	5.0	0.72	10.3
Nutbush (17)	-	0.69	0.7	0.20	1.08	8.2
Lamb (19)	O	0.60	0.03	0.44	0.03	4.0
Meander (21)	O	0.62	0.03	0.37	0.03	5.0
Mendota (22)	E	0.68	1.2	0.22	1.73	12

Table 24 (continued). US OECD DATA APPLIED TO DILLON'S PHOSPHORUS LOADING-PHOSPHORUS RETENTION AND MEAN DEPTH RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Retention Coefficient, R^b	Phosphorus Loading, L $(g/m^2/yr)^c$	Flushing Rate, ρ $(yr^{-1})^d$	$L(1-R)/\rho$ (mg/m^2)	Mean Depth, $\bar{z}$ (m)
Michigan (Open Waters)						
1971 (23-A)	O	0.84	0.14	0.03	0.72	84
1974 (24-A)	O	0.84	0.1	0.03	0.51	84
1971 (23-B)	O	0.91	0.14	0.01	1.27	
1974 (24-B)	O	0.91	0.10	0.01	0.91	
Lower Lake Minnetonka						
1969 (25)	E	0.72[f]	0.5	0.16	0.88	8.3
1973 (26)	E→M	0.72[f]	0.1 (0.2)[g]	0.16	0.18 (0.35)[g]	8.3
Potomac Estuary	U-E					
Upper (28)	-	0.17	85	25	2.83	4.8
Middle (29)	-	0.3	8	5.56	1.01	5.1
Lower (30)	-	0.48	1.2	1.18	0.53	7.2
Redstone (31)	E	0.46-0.50	1.44-1.68	1.0-1.4	0.68	4.3
Sallie (32)	E	0.51-0.57	1.5-4.2	0.56-0.91	1.78	6.4
Sammamish (33)	M	0.57	0.7	0.56	0.54	18
Shagawa (34)	E	0.47	0.7	1.25	0.30	5.7
Stewart (35)	E	0.22	4.82-8.05	12.5	0.40	1.9
Tahoe (36)	U-O	0.96	0.05	0.001	1.53	313
East Twin						
1972 (39)	E	0.47	0.7 (0.7)[g]	1.25	0.30 (0.30)[g]	5.0
1973 (40)	E	0.49	0.5 (0.5)	1.11	0.23 (0.23)	5.0
1974 (41)	E	0.41	0.7 (0.8)	2.0	0.21 (0.24)	5.0

Table 24 (continued). US OECD DATA APPLIED TO DILLON'S PHOSPHORUS LOADING-PHOSPHORUS RETENTION AND MEAN DEPTH RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Retention Coefficient, R[b]	Phosphorus Loading, L ($g/m^2/yr$)[c]	Flushing Rate, ρ (yr^{-1})[d]	$L(1-R)/\rho$ (mg/m^2)	Mean Depth, $\bar{z}$ (m)
West Twin						
1972 (43)	E	0.56	0.4 (0.4)[g]	0.62	0.28 (0.28)[g]	4.3
1973 (44)	E	0.57	0.3 (0.2)	0.56	0.23 (0.15)	4.3
1974 (45)	E	0.50	0.3 (0.3)	1.0	0.15 (0.15)	4.3
Twin Valley (46)	E	0.39-0.41	1.74-2.05	2.0-2.5	0.51	3.8
Virginia (47)	E	0.49-0.63	1.15-1.48	0.36-1.1	0.80	1.7
Waldo (48)	U-O	0.82	0.017	0.05	0.06	36
Washington						
1957 (49)	E	0.61	1.2	0.42	1.11	33
1964 (50)	E	0.61	2.3	0.42	2.14	33
1971 (51)	M	0.61	0.43	0.42	0.40	33
1974 (52)	M	0.61	0.47	0.42	0.44	33
Weir (53)	M	0.67	0.14	0.24	0.20	6.3
Wingra (54)	E	0.39	0.9	2.5	0.22	2.4

EXPLANATION:

[a]Investigator-indicated trophic state:

E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

Table 24 (continued). US OECD DATA APPLIED TO DILLON'S PHOSPHORUS LOADING-PHOSPHORUS RETENTION AND MEAN DEPTH RELATIONSHIP

EXPLANATION (continued)

[b]Retention coefficient, $R = 1/(1+\sqrt{\rho_\omega})$, where $\rho_\omega = 1/\tau_\omega$ = 1/hydraulic residence time (Vollenweider, 1975a; 1976a). See Table 20 for hydraulic residence time for US OECD water bodies.

[c]Based on investigator's estimates.

[d]Flushing rate, ρ = (discharge (m^3/yr)/water body volume (m^3) = $1/\tau_\omega$).

[e]Identification number for Figure 25 (See Table 14).

[f]Whole lake value.

[g]Data in parentheses represents data received by these reviewers from the principal investigators subsequent to completion of this report. Figure 25 is based on the original data reported by the investigators and does not reflect the revised data. Examination of the revised data indicated no significant changes in the overall conclusions concerning these water bodies.

US OECD PHOSPHORUS DATA APPLIED IN LARSEN AND MERCIER'S INFLUENT PHOSPHORUS AND PHOSPHORUS RETENTION RELATIONSHIP

Larsen and Mercier (1976) proposed another alternate method of examining the nutrient loading-trophic response relationships in water bodies. Consistent with the view that the phosphorus concentration in a water body, rather than the phosphorus loading to the water body, ultimately controls algal blooms and the eutrophication process (Vollenweider, 1968; Vollenweider and Dillon, 1974), Larsen and Mercier (1976) devised a phosphorus loading diagram which related a water body's trophic state to its influent phosphorus concentration, as modified by its phosphorus retention coefficient, R(P). They described the mean phosphorus concentration in a water body as the relationship between its mean influent phosphorus concentration and its ability to assimilate this influent phosphorus. The derivation of this approach was presented in an earlier section of this report. The Larsen-Mercier approach of utilizing the water body influent phosphorus concentrations rather than the phosphorus loading may be particularly important for water bodies that receive a substantial part of their key limiting nutrient load in a form that is not immediately available for aquatic plant growth. An example would be the phosphorus present in erosional material. In such cases, the phosphorus loading would not accurately predict the ultimate aquatic plant growth within the water body. As indicated earlier, Cowen et al. (1976a) have found that typically up to 20 percent of the nonsoluble orthophosphate present in US tributaries to Lake Ontario is available for algal growth in Lake Ontario.

Curves delineating trophic zones can be drawn on Larsen and Mercier's loading diagram, analogous to the trophic zones in the Vollenweider phosphorus loading diagram (Figure 19). The relative degree of eutrophy or oligotrophy of a water body is a function of its vertical displacement above or below the permissible phosphorus concentration line. The permissible and excessive phosphorus concentration lines correspond to the 10 μg/l and 20 μg/l limits determined by Sawyer (1947) and Sakamoto (1966), respectively. They are included in the loading diagram, according to Larsen and Mercier (1976), mainly for "illustrative purposes."

The Larsen and Mercier diagram, containing the US OECD water bodies, is presented in Figure 26. The pertinent US OECD data are presented in Table 25. If a data range was reported for a water body, the mean value was used in all calculations. Generally, the results of Figure 26 agree with those of Figures 22 and 25. In most cases, the predicted trophic states are in agreement with those reported by the US OECD investigators. A feature of Larsen and Mercier's relationship is that it allows one to relate the mean phosphorus concentration of a water body to both its phosphorus loading and its mean influent phosphorus concentration. If two of the above parameters are known, one can use the interrelationship between the three components to determine the value of the third parameter.

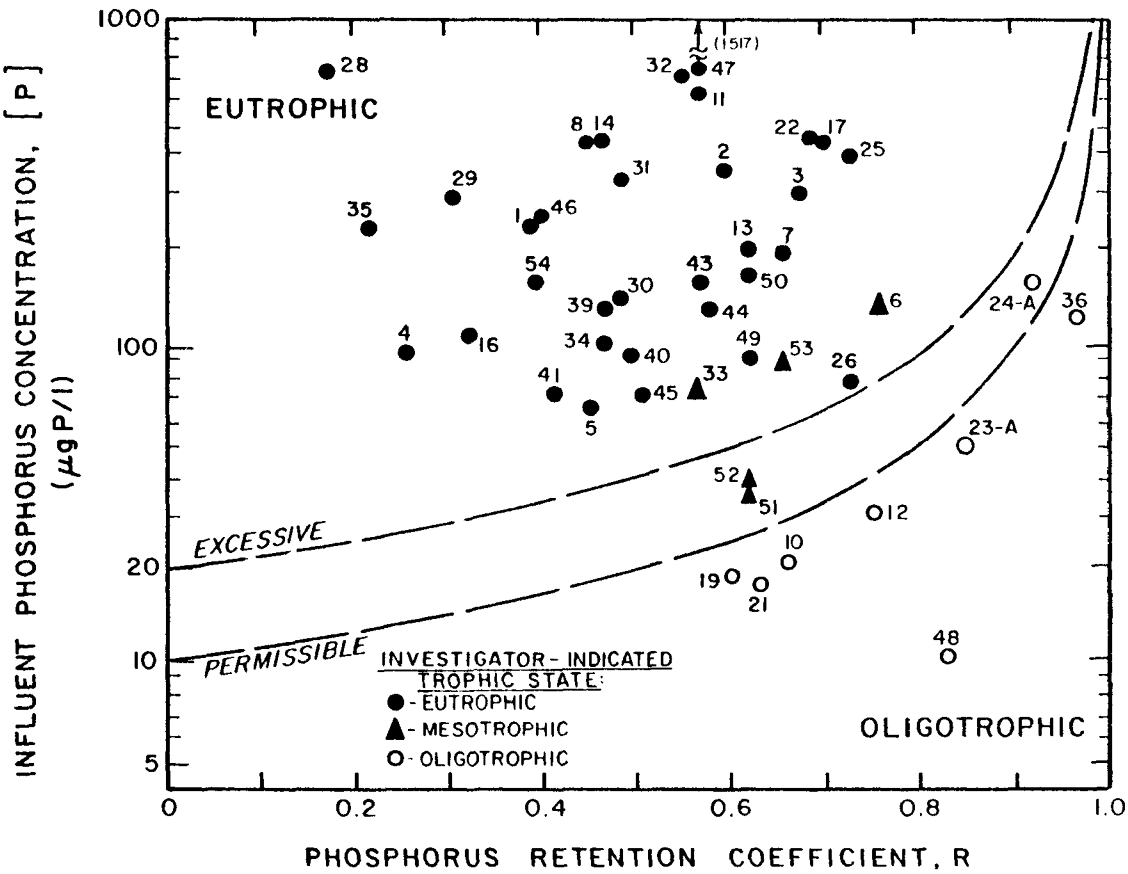

Figure 26. US OECD Data Applied to Larsen and Mercier Influent Phosphorus and Phosphorus Retention Relationship

Table 25. US OECD DATA APPLIED TO LARSEN AND MERCIER'S INFLUENT PHOSPHORUS CONCENTRATION AND PHOSPHORUS RETENTION RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Retention Coefficient, R[b]	Influent Phosphorus Concentration, $\overline{[P]}$ (μg/l)[c]
Blackhawk (1)[d]	E	0.41	227
Brownie (2)	E	0.59	347
Calhoun (3)	E	0.66	297
Camelot-Sherwood (4)	E	0.25	92.0
Canadarago (5)	E	0.44	62.4
Cayuga (6)	M	0.75	127
Cedar (7)	E	0.64	194
Cox Hollow (8)	E	0.44	285
Dogfish (10)	O	0.65	18.2
Dutch Hollow (11)	E	0.57	576
George (12)	O-M	0.74	32
Harriet (13)	E	0.61	192
Isles (14)	E	0.44	451
Kerr Reservoir	E-M		
Roanoke (16)	-	0.31	101
Nutbush (17)	-	0.69	438
Lamb (19)	O	0.60	17.6
Meander (21)	O	0.62	16.7
Mendota (22)	E	0.68	444
Michigan (Open Waters)			
(23-A)	O	0.84	50
(24-A)	O	0.84	36
(23-B)	O	0.91	167
(24-B)	O	0.91	119
Lower Lake Minnetonka			
1969 (25)	E	0.72[e]	417
1973 (26)	E→M	0.72[e]	76.9 (138)[f]
Potomac Estuary	U-E		
Upper (28)	-	0.17	708
Middle (29)	-	0.30	283
Lower (30)	-	0.48	142
Redstone (31)	E	0.48	300

Table 25 (continued). US OECD DATA APPLIED TO LARSEN AND MERCIER'S INFLUENT PHOSPHORUS CONCENTRATION AND PHOSPHORUS RETENTION RELATIONSHIP

Water Body	Trophic State[a]	Phosphorus Retention Coefficient, R[b]	Influent Phosphorus Concentration, $\overline{[P]}$ (μg/l)[c]
Sallie (32)	E	0.54	606
Sammamish (33)	M	0.57	70
Shagawa (34)	E	0.47	98.6
Stewart (35)	E	0.22	270
Tahoe (36)	U-O	0.96	111
East Twin			
1972 (39)	E	0.47	113 (113)[f]
1973 (40)	E	0.49	89.3 (89.3)
1974 (41)	E	0.41	70.0 (80.0)
West Twin			
1972 (43)	E	0.56	148 (148)[f]
1973 (44)	E	0.57	125 (83.8)
1974 (45)	E	0.50	69.8 (69.8)
Twin Valley (46)	E	0.40	222
Virginia (47)	E	0.56	1052
Waldo (48)	U-O	0.82	10.1
Washington			
1957 (49)	E	0.61[e]	87.0
1964 (50)	E	0.61[e]	167
1971 (51)	M	0.61[e]	31.2
1974 (52)	M	0.61[e]	34.0
Weir (53)	M	0.67	46.7
Wingra (54)	E	0.39	150

EXPLANATION:

[a]Investigator-indicated trophic state:

E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

Table 25 (continued). US OECD DATA APPLIED TO LARSEN AND MERCIER'S INFLUENT PHOSPHORUS CONCENTRATION AND PHOSPHORUS RETENTION RELATIONSHIP

EXPLANATION (continued).

[b]Retention coefficient, $R = 1/(1+\sqrt{\rho_\omega})$, where $\rho_\omega = 1/\tau_\omega$ = 1/hydraulic residence time (Vollenweider, 1975a; 1976a). See Table 20 for hydraulic residence times of US OECD water bodies.

[c]Mean influent phosphorus concentration, $\overline{[P]} = L(P)/q_s$, where $L(P)$ = phosphorus loading ($mg/m^2/yr$) and q_s = hydraulic loading = $\bar{z}/\tau_\omega$, where $\bar{z}$ = mean depth (m) and τ_ω = hydraulic residence time. See Table 15 for influent phosphorus concentrations for US OECD water bodies.

[d]Identification number for Figure 26 (see Table 14).

[e]Whole lake value.

[f]All data in parentheses represents data submitted to these reviewers from the principal investigators subsequent to the completion of this report. Figure 26 is based on the original data submitted by the investigators and does not reflect the revised data. Examination of the revised data indicated no significant changes in the overall conclusions concerning these water bodies.

In summary, the results of Vollenweider's phosphorus loading characteristics and mean chlorophyll a concentration relationship (Figure 22), Dillon's phosphorus loading/phosphorus retention and mean depth relationship (Figure 25) and Larsen and Mercier's influent phosphorus concentration and phosphorus retention relationship (Figure 26), all either directly or indirectly support Vollenweider's approach for estimating critical phosphorus loads for lakes and impoundments. Furthermore, they generally support the possible errors in the phosphorus loading estimates suggested in Figures 14 and 15. This supports both the validity of the Vollenweider relationship illustrated in Equation 26, and the use of watershed land use nutrient export coefficients as methods of estimating phosphorus loadings and of checking the reasonableness of calculated phosphorus loadings. Finally, these three models offer a certain capacity, based on the phosphorus loadings, for predicting the mean phosphorus and mean chlorophyll a concentrations in a water body.

SECTION X

CORRELATIONS BETWEEN NUTRIENT LOADINGS AND EUTROPHICATION RESPONSE PARAMETERS

This section of this report is devoted to analysis of the correlations between the nutrient loading for the US OECD water bodies and their eutrophication response to these loadings. A list of suggested correlations was developed by R. Vollenweider and members of the OECD Eutrophication Technical Bureau and was distributed to the OECD eutrophication principal investigators. Many of these suggested correlations could not be made for the lakes in the US OECD eutrophication study since only a limited number of investigators had data for all of the parameters required to make these correlations. Included in the list of suggested eutrophication response parameters were maximum rates of primary production and respiration, stratified period average chlorophyll a content, average epilimnetic concentration of particulate phosphorus, areal hypolimnetic oxygen deficit, maximum oxygen surplus, duration of algal blooms and maximum rate of development of bloom. These data were not reported for the US OECD water bodies. In some instances, insufficient data were available to prepare a potentially meaningful plot of the data. For some parameters, the correlations have been presented and discussed in previous sections of this report. This section of this report presents what might be considered miscellaneous correlations which are thought to be of lesser importance than those presented in other parts of the report or where there were insufficient data to justify a more intensive discussion of the relationship. A listing of the various correlations analyzed in this report is presented in Table 26.

Before presenting the results of these correlations between nutrient loadings and eutrophication response parameters, the reader should be made aware of several factors which limit the values of these analyses. First, as indicated in an earlier section of this report (Table 11), the various response parameters (i.e., nutrient concentrations) were measured using a variety of analytical techniques. In addition to differing analytical procedures, the sampling methodologies also varied widely, which could affect the results obtained for a given response parameter measurement. As indicated in the Summary Sheets (Appendix II), the US OECD water bodies were sampled at a variety of depths and locations and on differing dates. For example, some water bodies were sampled frequently all year, others were sampled frequently part of the year and less frequently the rest of the year, while still

Table 26. LIST OF CORRELATIONS EXAMINED IN US OECD WATER BODIES[a]

I. Phosphorus Loading and:

- A. annual mean chlorophyll a (Figure 27);
- B. annual mean Secchi depth (Figure 28);
- C. annual mean total phosphorus (Figure 29);
- D. annual mean dissolved phosphorus (Figure 30);
- E. annual primary productivity (Figure 31);
- F. annual total primary production (Figure 32);
- G. growing season epilimnetic chlorophyll a (Figure 33);
- H. growing season epilimnetic total phosphorus (Figure 34);
- I. growing season epilimnetic dissolved phosphorus (Figure 35);
- J. growing season epilimnetic primary productivity (Figure 36);
- K. spring overturn total phosphorus (Figure 37);
- L. spring overturn dissolved phosphorus*

II. Nitrogen Loading and:

- A. annual mean chlorophyll a (Figure 38);
- B. annual mean Secchi depth (Figure 39);
- C. annual mean inorganic nitrogen (Figure 40);
- D. annual primary productivity (Figure 41);
- E. annual total primary production (Figure 42);
- F. growing season epilimnetic chlorophyll a (Figure 43);
- G. growing season epilimnetic inorganic nitrogen (Figure 44);
- H. growing season epilimnetic primary productivity (Figure 45);
- I. spring overturn inorganic nitrogen (Figure 46).

III. Annual Mean Total Phosphorus and:

- A. annual mean chlorophyll a (Figure 47);
- B. annual mean Secchi depth (Figure 48);

Table 26 (continued). LIST OF CORRELATIONS EXAMINED IN US OECD WATER BODIES

C. annual mean dissolved phosphorus (Figure 49);

D. annual primary productivity (Figure 50);

E. growing season epilimnetic chlorophyll a (Figure 51);

F. growing season epilimnetic primary productivity (Figure 52);

G. spring overturn total phosphorus (Figure 53).

IV. Growing Season Epilimnetic Total Phosphorus and:

A. growing season epilimnetic chlorophyll a (Figure 54);

B. growing season epilimnetic primary productivity (Figure 55).

V. Spring Overturn Total Phosphorus and:

A. growing season epilimnetic chlorophyll a (Figure 56);

B. growing season epilimnetic total phosphorus (Figure 57);

C. growing season epilimnetic dissolved phosphorus (Figure 58).

VI. Annual Mean Dissolved Phosphorus and:

A. annual mean chlorophyll a (Figure 59);

B. annual primary productivity (Figure 60);

C. spring overturn dissolved phosphorus (Figure 61).

VII. Growing Season Epilimnetic Dissolved Phosphorus and:

A. growing season epilimnetic chlorophyll a (Figure 62).

VIII. Spring Overturn Dissolved Phosphorus and:

A. growing season epilimnetic chlorophyll a (Figure 63);

B. growing season epilimnetic dissolved phosphorus (Figure 64);

C. growing season epilimnetic primary productivity*

Table 26 (continued). LIST OF CORRELATIONS EXAMINED IN US OECD WATER BODIES

IX. Annual Mean Inorganic Nitrogen and:

A. annual mean chlorophyll a (Figure 65);

B. annual mean Secchi depth (Figure 66);

C. annual primary productivity (Figure 67);

D. growing season epilimnetic chlorophyll a (Figure 68);

E. growing season epilimnetic primary productivity (Figure 69);

F. spring overturn inorganic nitrogen*

X. Growing Season Epilimnetic Inorganic Nitrogen and:

A. growing season epilimetic chlorophyll a (Figure 70)

B. growing season epilimnetic primary productivity (Figure 71).

XI. Others:

A. annual primary productivity and annual mean chlorophyll a (Figure 72);

B. annual mean chlorophyll a and annual mean Secchi depth (see Figures 77 and 78);

C. annual primary productivity and mean Secchi depth (Figure 73);

D. growing season mean primary productivity and growing season mean chlorophyll a (Figure 74);

E. annual mean daily primary productivity and annual mean chlorophyll a (Figure 75);

F. annual mean daily primary productivity and annual mean areal chlorophyll a (Figure 75).

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[a]Data taken from Summary Sheets (Appendix II).

*Insufficient data available.

others were sampled infrequently all year. Also, some reported mean values were arithmetic means of several sampling depths, others were mean values integrated over the sampling depths, while still others were surface or epilimnetic mean values. As discussed in an earlier section, these factors can all contribute to an erroneous "mean" value for a given response parameter measurement. It is not possible to determine the extent of possible errors in the parameters used in the correlations. This section presents a general idea of the correlation(s) that may exist between nutrient loads and eutrophication response parameters in the US OECD water bodies. No statistical evaluation of the correlation data was undertaken. This report is limited to a simple visual examination of the correlations in a graphical form for obvious trends. A 'correlation' as used in this section of the report indicates that a relationship, either positive or negative, appears to exist between two parameters on the basis of a visual inspection of a plot of these two parameters. No attempt is made in these plots to indicate the particular water body responsible for the data. All data used in these plots are available in Appendix II. For some plots, the investigator-indicated trophic status is presented. For others, where there are obvious differences in the types of data for some parameters, this is also indicated on the plot.

PHOSPHORUS LOADINGS

Although there is a large amount of scattering of the data, there is a correlation observed between phosphorus loading and mean chlorophyll a (Figure 27). The scatter in this diagram, as well as all other correlations examined in this section, is partly due to sampling and analysis variability, as indicated earlier. In addition, the 'mean' chlorophyll a consisted of annual means, summer means and annual mean chlorophyll in the upper two meters of the water column. As algal growth is dependent on the loading, the correlation is expected. However, there usually is no clear correlation between phosphorus loading and the resulting algal biomass (as indicated by chlorophyll a content) in a water body (Vollenweider, 1968; Vollenweider and Dillon, 1974). It depends on a number of factors discussed earlier, such as the mean depth of the water body and its hydraulic residence time. Consequently, Figure 22, which incorporates the phosphorus loading to a water body, as modified by its assimilative capacity (i.e., $(L(P)/q_s)/(1+\sqrt{\tau_\omega})$, is a much better indicator of the phosphorus loading-chlorophyll response of a water body. Vollenweider (1976a) has shown a good correlation between these two parameters. The US OECD water bodies also show a good correlation (Figure 22).

There is a correlation between phosphorus loading and mean Secchi depth (Figure 28). The relationship is a negative hyperbolic function on this semi-log plot, although it exhibits a certain degree of scatter. A negative relationship between Secchi depth and chlorophyll a has been reported by Edmondson (1972) and Carlson (1974). Since phosphorus loading is correlated with

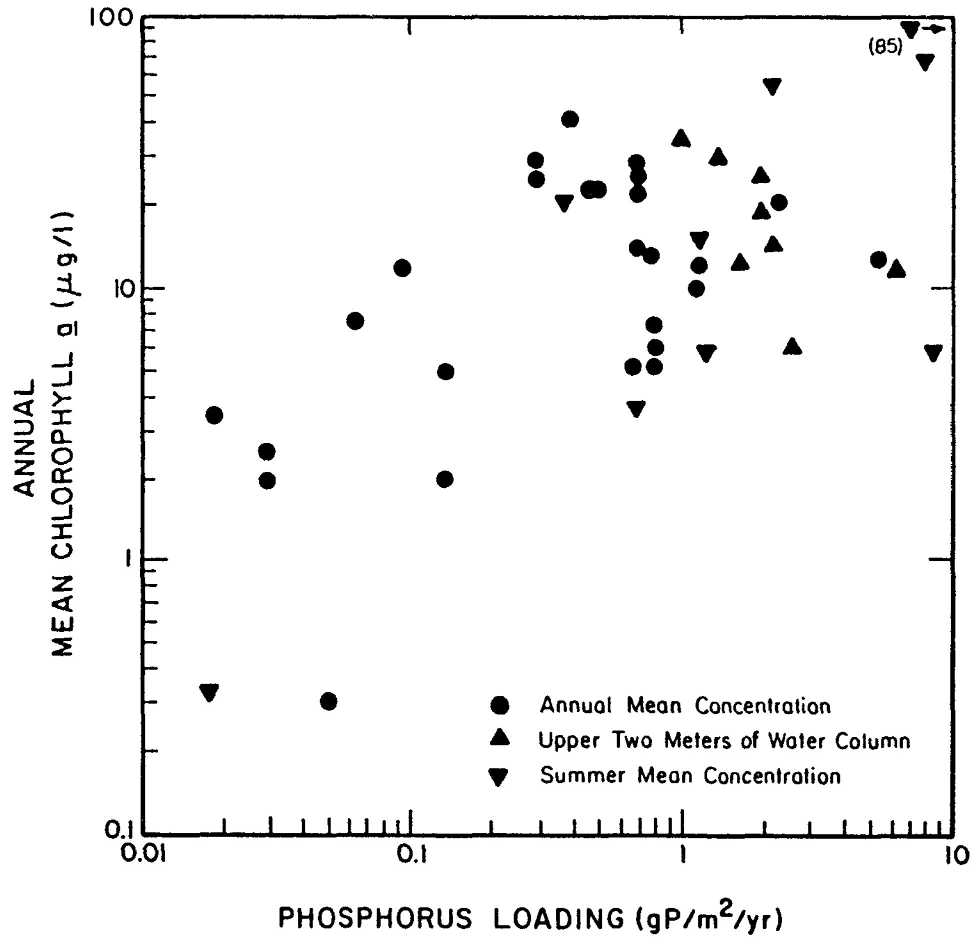

Figure 27. Phosphorus Loading and Mean Chlorophyll a Relationship in US OECD Water Bodies

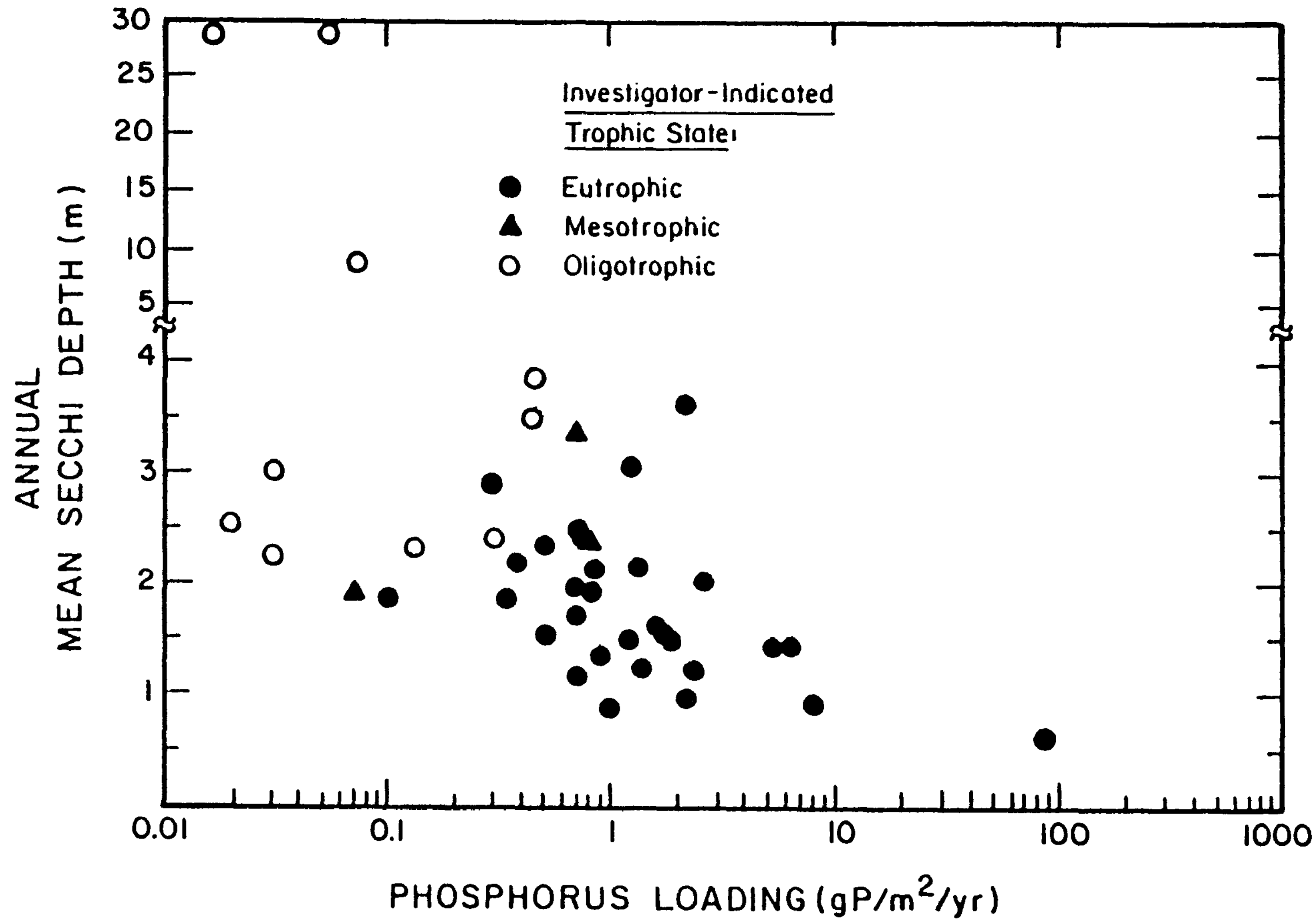

Figure 28. Phosphorus Loading and Mean Secchi Depth Relationship in US OECD Water Bodies

chlorophyll (Figure 22) and chlorophyll is correlated with Secchi depth, then a correlation should, and does, exist between phosphorus loading and Secchi depth. This relationship will be used in a following section of this report to indicate how changes in water quality can be related to changes in phosphorus loadings to a water body.

A positive correlation exists between phosphorus loading and mean total phosphorus in the water body (Figure 29). Although there is considerable data scatter, this correlation is not unexpected since the total phosphorus content of a water body will usually be a function of the input phosphorus. Contrastingly, there is not a readily observable correlation between phosphorus loading and the mean dissolved phosphorus concentrations in the US OECD water bodies (Figure 30), in view of the considerable scatter of the data. This lack of correlation is expected since dissolved phosphorus is the algal-available phosphorus form and will be readily assimilated by the algal population in a water body. It is expected that, in general, the available nutrients, both phosphorus and nitrogen, will not show a good correlation with any of the parameters examined in this section. The available nutrient concentration will increase and decrease in a water body, depending on the algal growth dynamics which fluctuate considerably during the annual cycle.

There appears to be a positive correlation between areal phosphorus loading and mean annual primary productivity (Figure 31) although the data are scattered and limited. In general, correlations between primary productivity and both nutrient loadings and concentrations, although usually present, were marked by considerable data scatter. This rendered this eutrophication response parameter of limited value. In addition, the question of macrophyte and attached algal primary production was not addressed in this study. In contrast with primary productivity, there is no readily observable correlation between phosphorus loading and total primary production (i.e., g C/yr in the water body) in the US OECD water bodies (Figure 32). The total production, as a function of phosphorus loading, appears to vary widely.

A positive correlation appears to exist between phosphorus loadings and the growing season epilimnetic concentrations of both chlorophyll _a_ and total phosphorus (Figures 33 and 34, respectively). (Note: the growing season, as used in this report, was the period between May and October. However, the growing season varied considerably between water bodies, being less for some water bodies and considerably longer for others such as the Kerr Reservoir and Lake Weir. Since such differences in growing season could not be standardized, all "growing season" values, regardless of length of growing season, were assumed to be equivalent in the correlations). By contrast, there is no correlation between phosphorus loading and the growing season dissolved phosphorus concentration (Figure 35). As indicated above, this is not unexpected since the dissolved phosphorus concentrations will vary as a function of algal growth, rather than

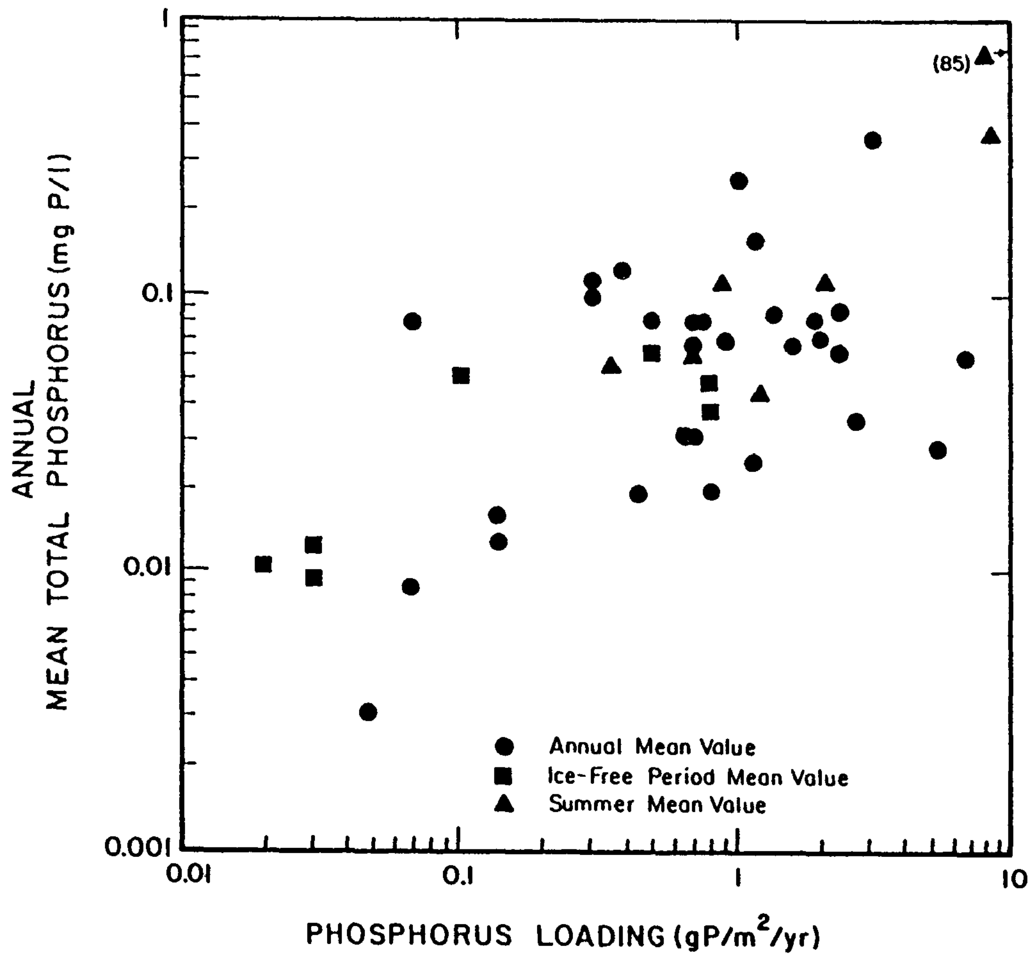

Figure 29. Phosphorus Loading and Mean Total Phosphorus Relationship in US OECD Water Bodies

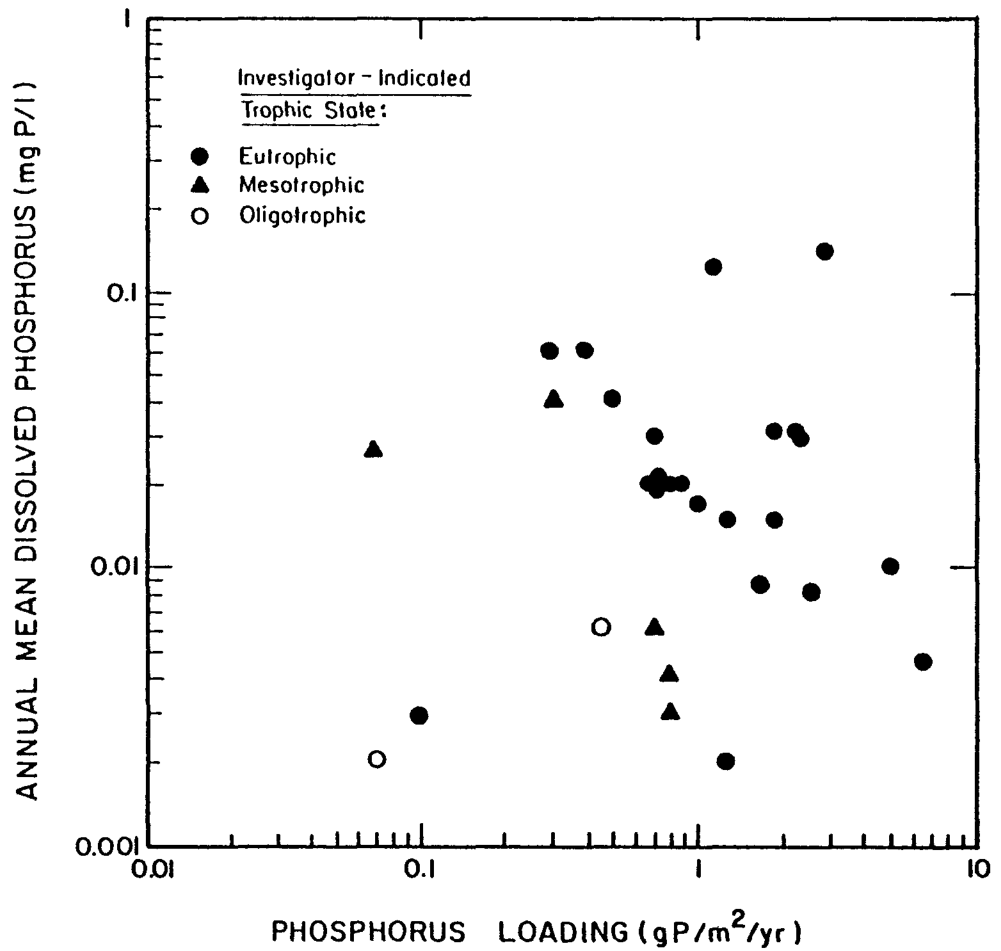

Figure 30. Phosphorus Loading and Mean Dissolved Phosphorus Relationship in US OECD Water Bodies

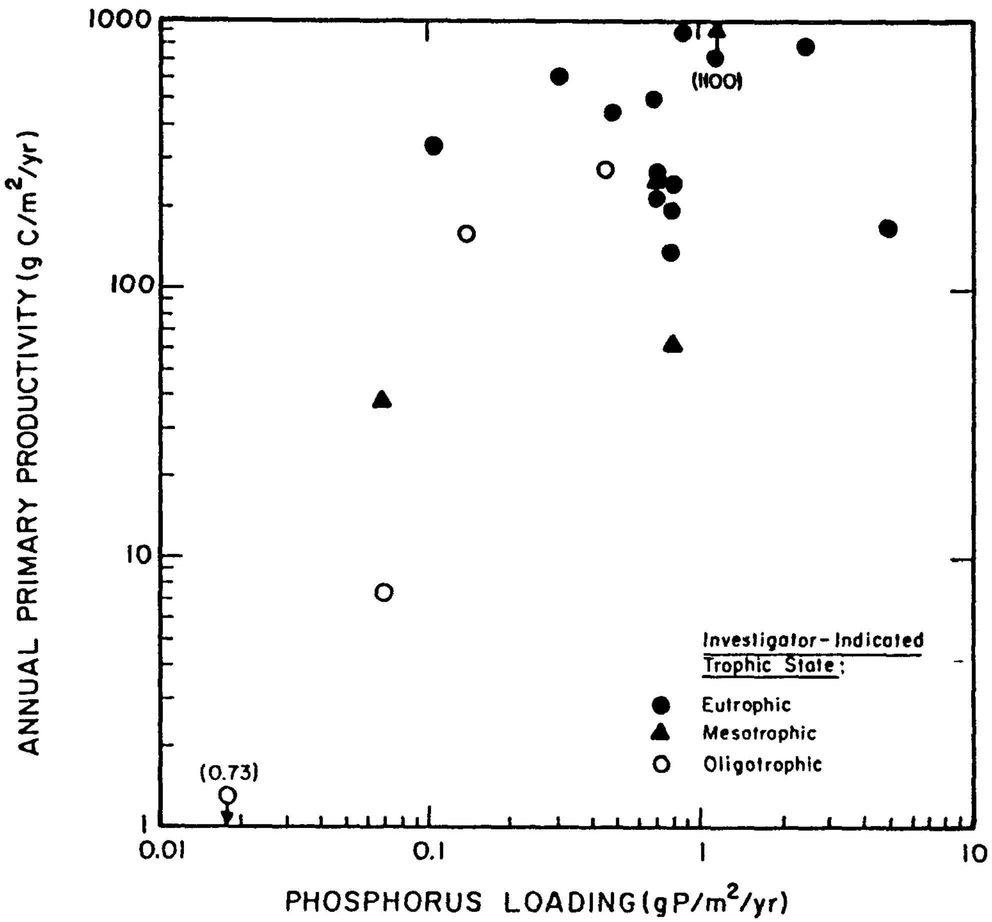

Figure 31. Phosphorus Loading and Primary Productivity Relationship in US OECD Water Bodies

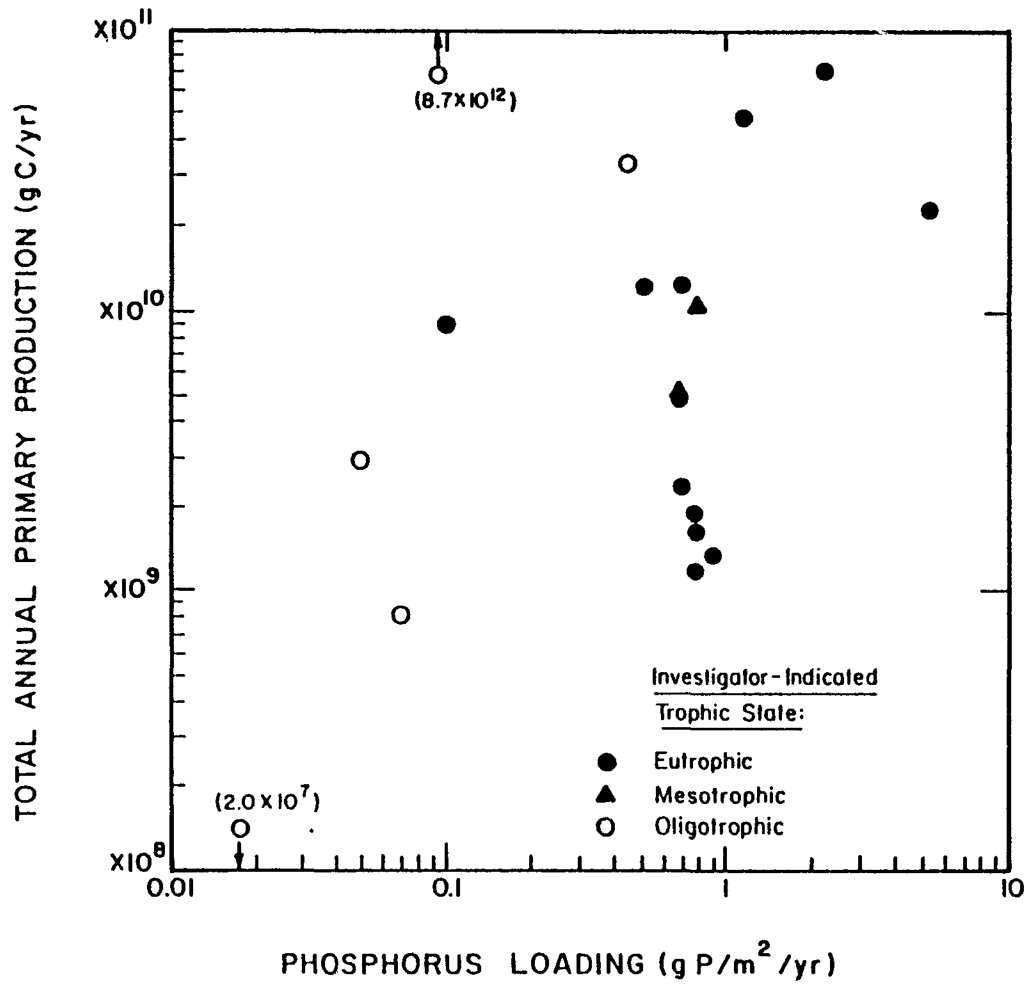

Figure 32. Phosphorus Loading and Total Primary Production Relationship in US OECD Water Bodies

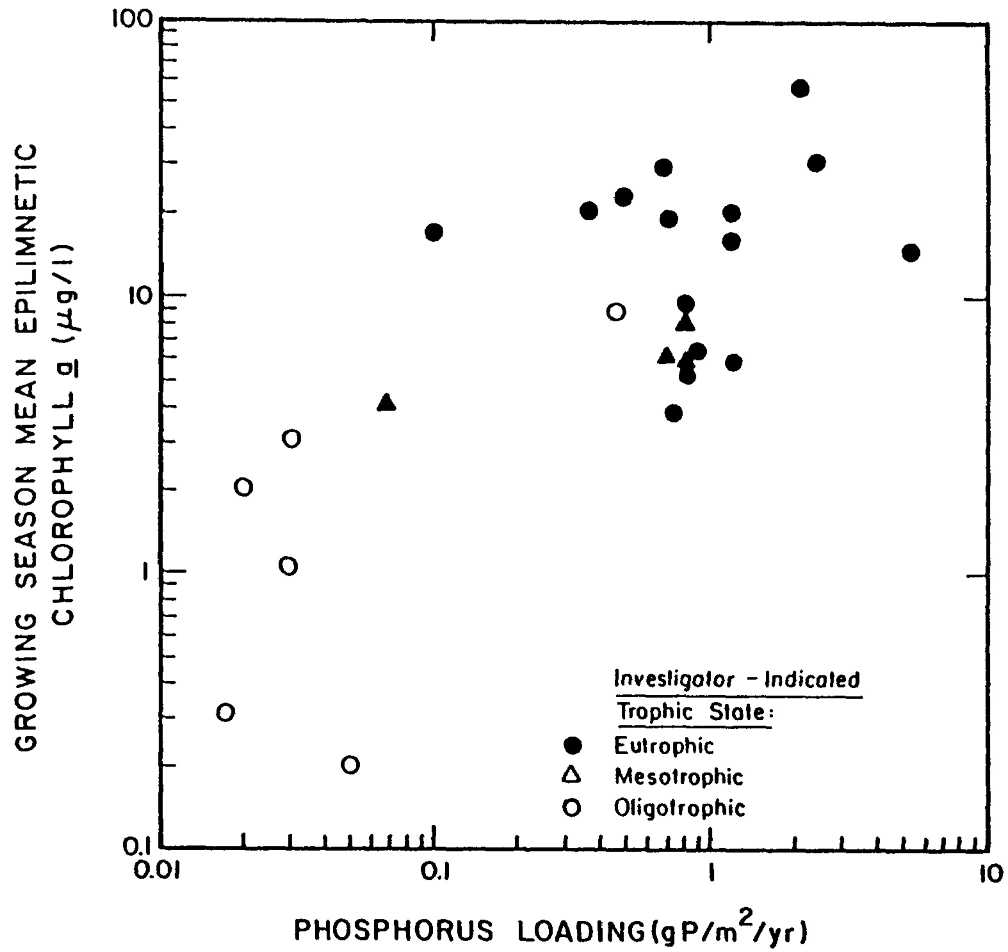

Figure 33. Phosphorus Loading and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

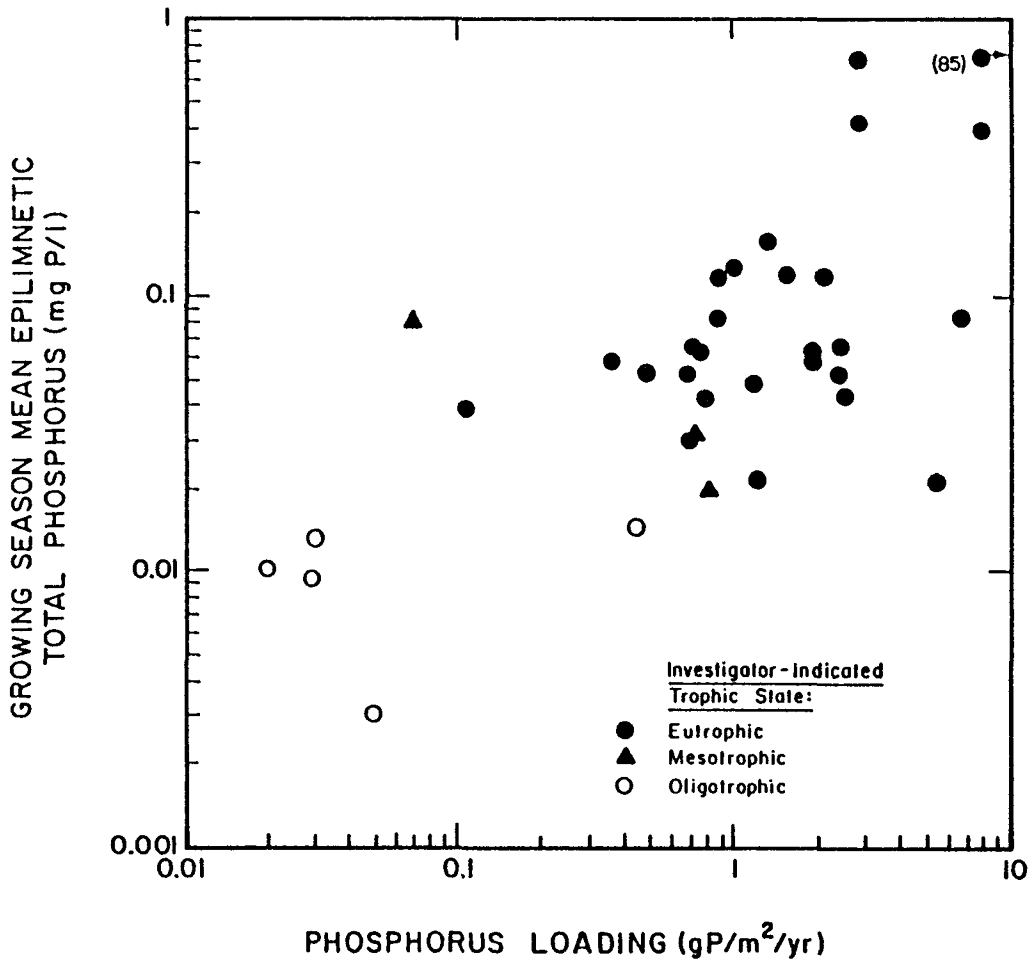

Figure 34. Phosphorus Loading and Growing Season Epilimnetic Total Phosphorus Relationship in US OECD Water Bodies

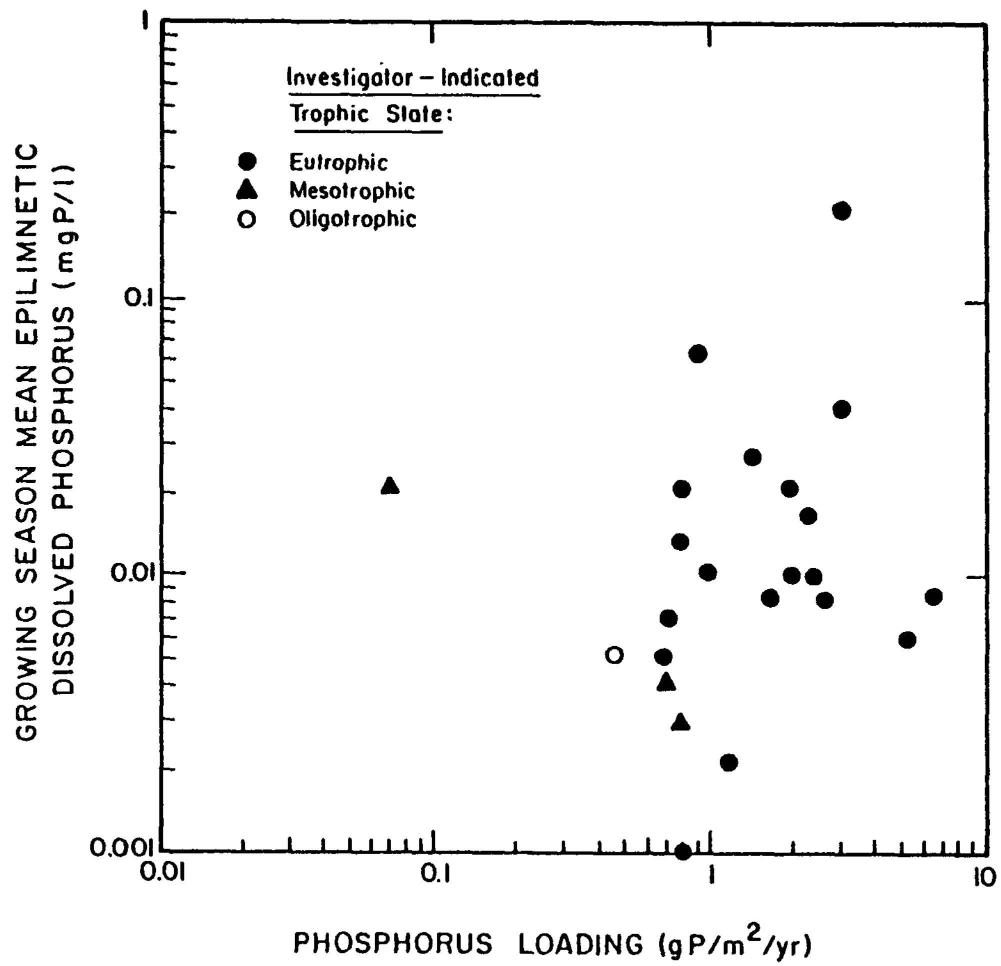

Figure 35. Phosphorus Loading and Growing Season Epilimnetic Dissolved Phosphorus Relationship in US OECD Water Bodies

phosphorus supply. It is not clear whether there is a correlation between phosphorus loading and growing season primary productivity (Figure 36) mainly because of scarcity of data. The growing season primary productivity was not measured in most US OECD water bodies. There appears to be a poor correlation between phosphorus loading and the spring overturn total phosphorus (Figure 37), although there is also a scarcity of data for this correlation. This is somewhat surprising since the total phosphorus throughout the year should generally be a function of the input phosphorus. There are not sufficient data available to examine the correlation between phosphorus loading and spring overturn dissolved phosphorus. A reasonably good correlation should be found for these two parameters for lakes which normally have ice cover during the winter.

NITROGEN LOADINGS

It should be noted before examining the correlations between nitrogen loadings and eutrophication response parameters that most of the US OECD water bodies are phosphorus-limited (Table 9) with respect to algal growth requirements. Nitrogen loadings were not reported for a number of the US OECD water bodies with the result that the US OECD data base for nitrogen loads is less extensive than that for phosphorus loads. The application of any of the correlations in this section for providing justification for a certain type of eutrophication control measure should be made with caution.

A positive correlation was found between nitrogen loading and mean chlorophyll a (Figure 38). The correlation is very similar to that seen between phosphorus loading and chlorophyll a (Figure 26). There is an order of magnitude increase on the loading axis of the graph, but the relative positions of the water bodies are similar. This illustrates the relatively constant input of nitrogen relative to phosphorus. This is consistent with Vollenweider's (1968) use of a 15N:1P loading ratio (by weight) in his original loading diagrams (Figures 5 and 6). Since most of the US OECD water bodies are phosphorus-limited, one must view the positive correlation between nitrogen loading and mean chlorophyll a with caution. The relatively constant N:P loading ratio may be producing an artifact with respect to this relationship. This possibility is illustrated in examination of the correlation between nitrogen loadings and Secchi depth (Figure 39). Although there are fewer data points than with phosphorus loads, there is a considerable amount of data scatter in this relationship (i.e., a nitrogen load of approximately 2 g $N/m^2/yr$ producing a Secchi depth range of about 2 to 9 meters, to cite one example). This would suggest that the nitrogen load has less effect on the algal populations, and hence resultant Secchi depth, than does the phosphorus load. This view is consistent with a phosphorus-limitation of most US OECD water bodies.

A high positive correlation is found between nitrogen loading and mean inorganic nitrogen (Figure 40). The correlation

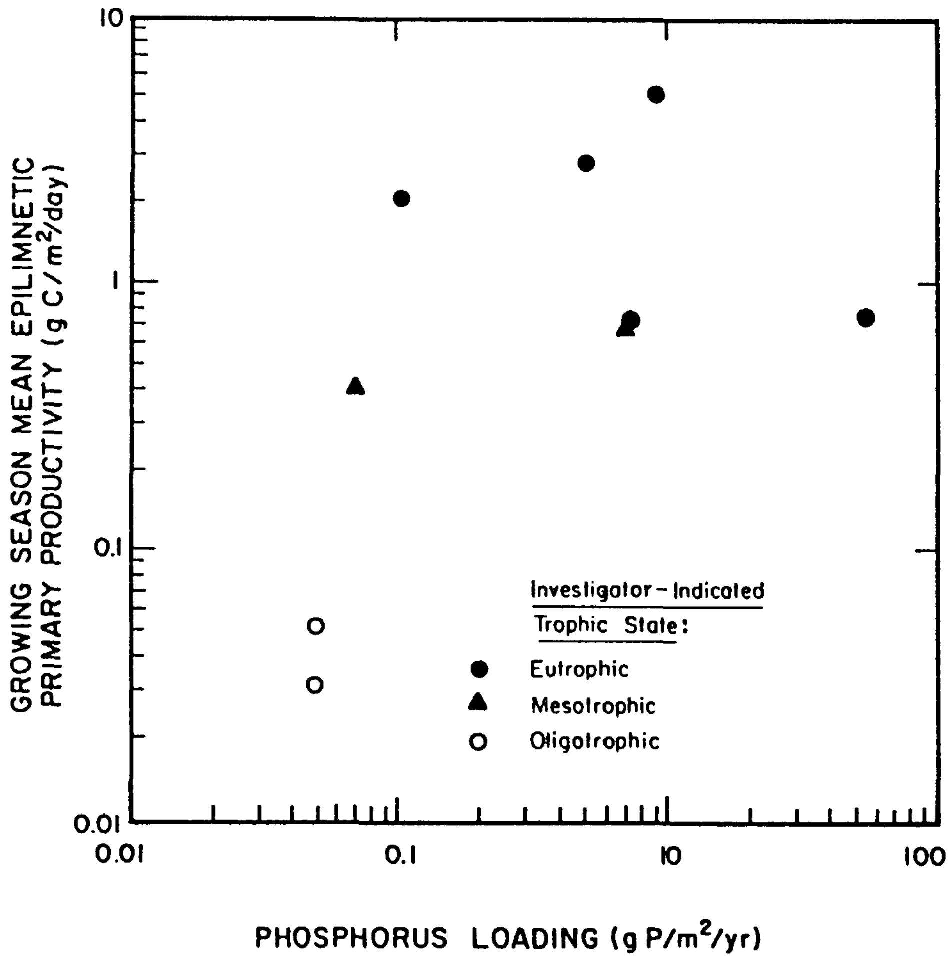

Figure 36. Phosphorus Loading and Growing Season Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

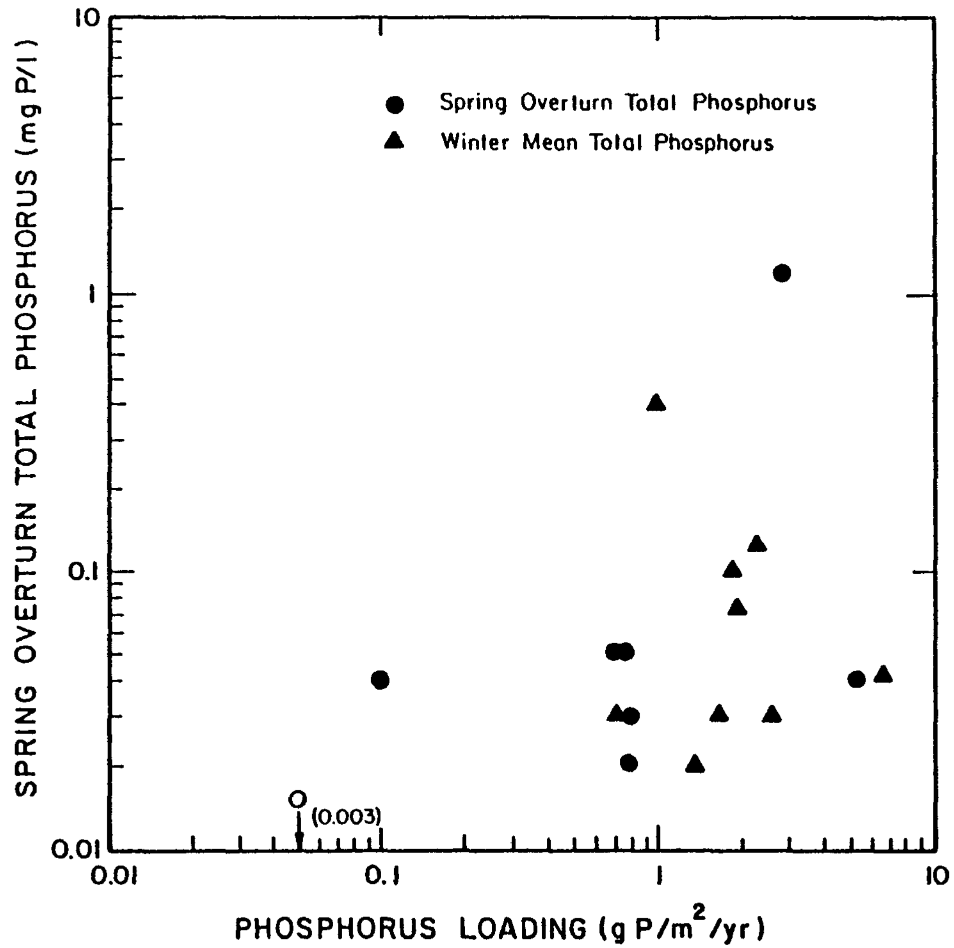

Figure 37. Phosphorus Loading and Spring Overturn Total Phosphorus Relationship in US OECD Water Bodies

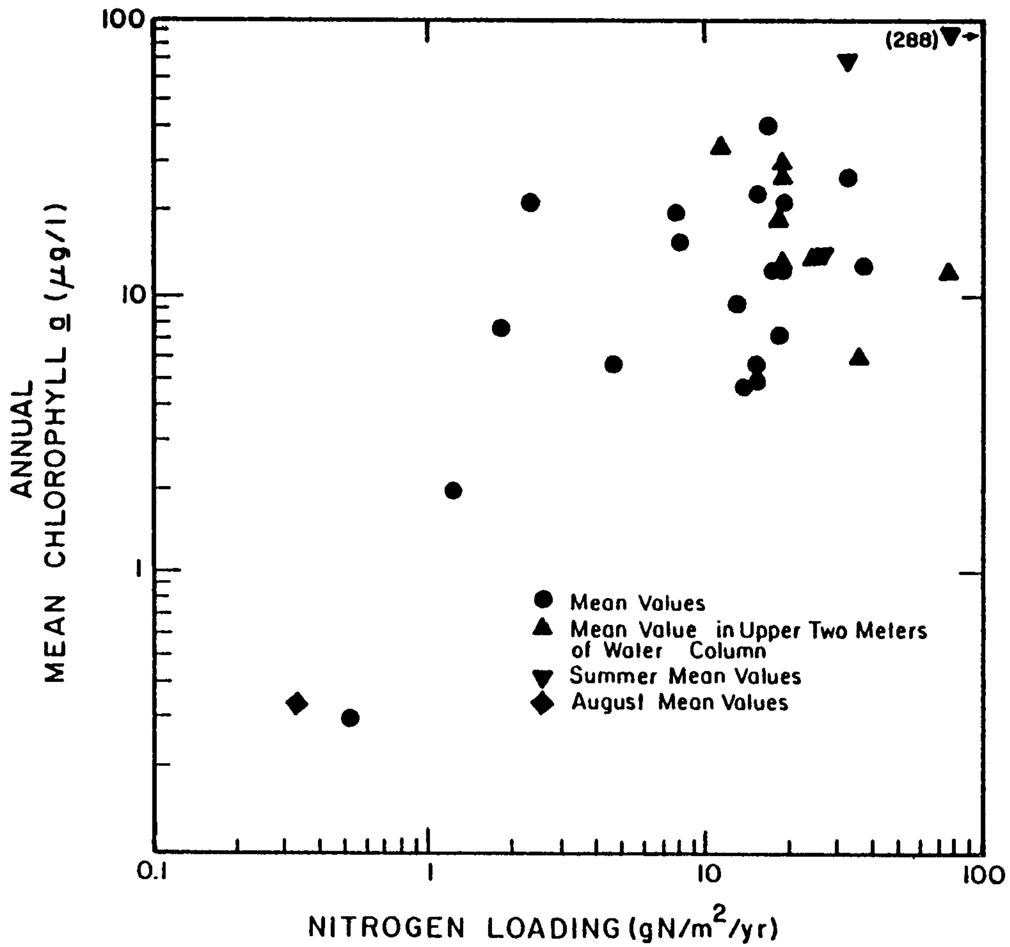

Figure 38. Nitrogen Loading and Mean Chlorophyll a Relationship in US OECD Water Bodies

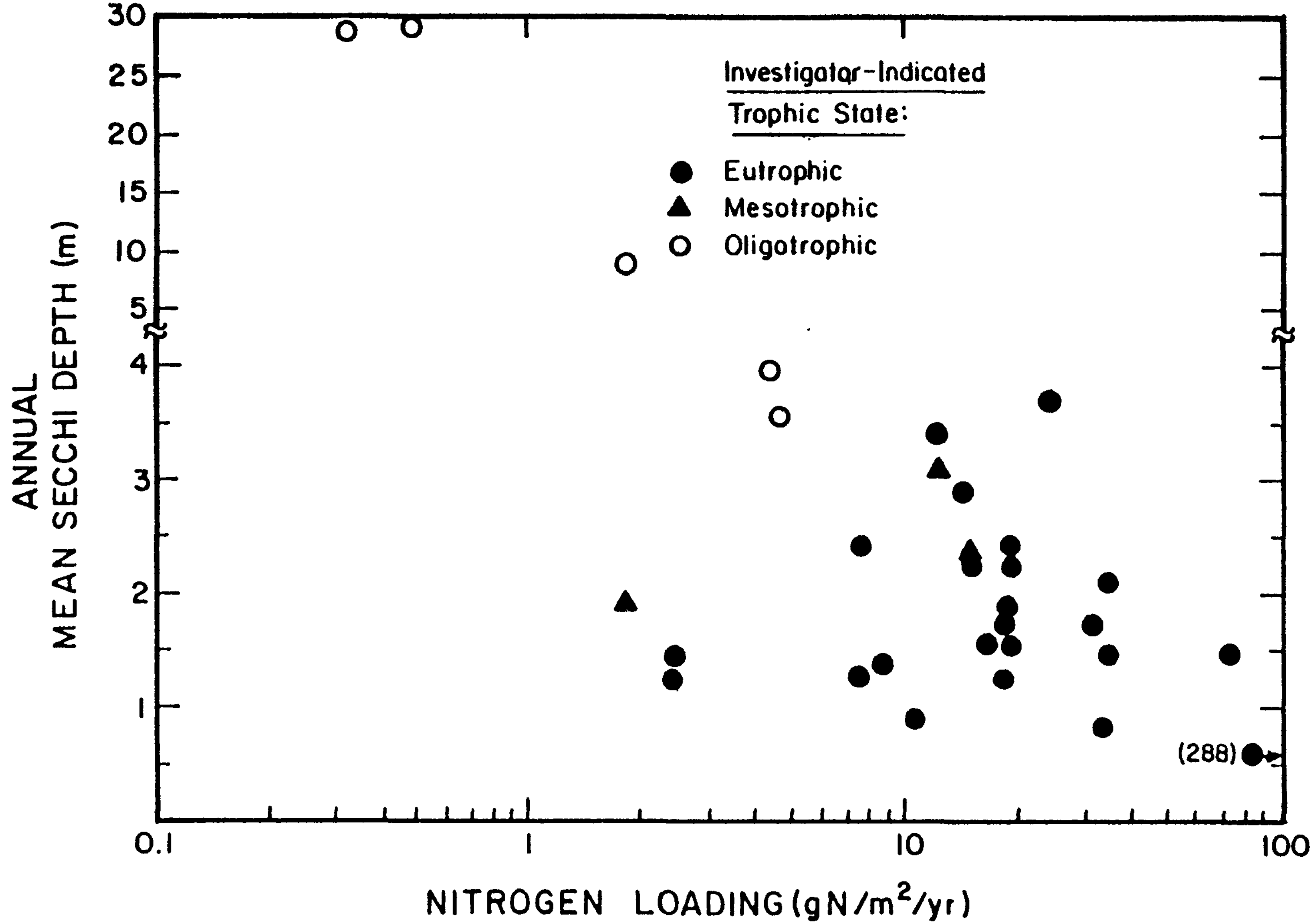

Figure 39. Nitrogen Loading and Mean Secchi Depth Relationship in US OECD Water Bodies

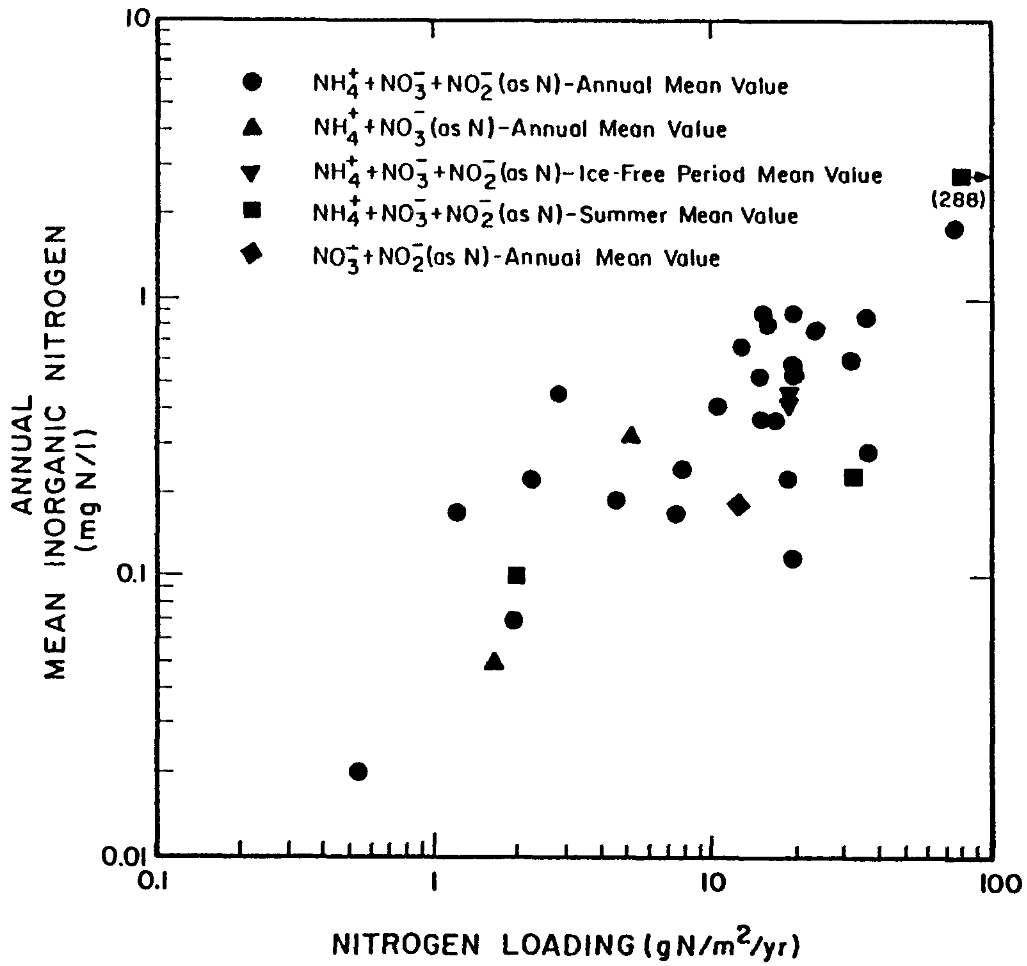

Figure 40. Nitrogen Loading and Mean Inorganic Nitrogen Relationship in US OECD Water Bodies

is better than that found between phosphorus load and either mean total phosphorus or mean dissolved phosphorus (Figure 29 and 30). This strong positive nitrogen loading-mean inorganic nitrogen correlation lends support to the view that most US OECD water bodies are phosphorus-limited, rather than nitrogen-limited. This high correlation indicates that the algal populations are not in general depleting the input nitrogen, regardless of the magnitude of the input. Rather, the inorganic nitrogen (i.e., algal-available nitrogen) is increasing as the loading is increasing. Thus, the algae are not growing in response to the input nitrogen, but rather in response to another nutrient. The lack of correlation between phosphorus loading and mean dissolved phosphorus (Figure 30) indicates the controlling nutrient is likely phosphorus.

There appears to be a positive correlation between nitrogen loading and primary productivity (Figure 41). The correlation appears to be about the same degree as that between phosphorus loading and primary productivity (Figure 31). However, there are fewer data sets for nitrogen loading than for phosphorus loading. Thus, this nitrogen load-primary productivity correlation may also be a coincidental artifact of the relatively constant N:P loading ratio found with the US OECD water bodies. There appears to be no readily observable correlation between total annual primary production and nitrogen loading (Figure 41). The data scatter is of the same magnitude as that between phosphorus loading and total annual primary production (Figure 32). This further illustrates the limited applicability of correlations between nutrient loads and both primary productivity and total production. It indicates the relationship between these parameters may be more complex than can be visualized using this single graphing technique.

It is difficult to determine if there is a correlation between nitrogen loading and growing season epilimnetic chlorophyll *a* (Figure 43). The correlation may be real, but the scarcity of data for these two parameters does not allow an accurate evaluation. For the common water bodies, the data scatter between these two parameters appears to be as great as that seen between phosphorus loading and growing season epilimnetic chlorophyll *a* (Figure 33). There is a better correlation between nitrogen loading and growing season epilimnetic inorganic nitrogen (Figure 44) than between phosphorus loading and either growing season epilimnetic total phosphorus or dissolved phosphorus (Figures 29 and 30, respectively). Thus, the growing season and annual mean algal-available nitrogen both seem to correlate reasonably well with their input. This growing season correlation (i.e., dependence) of inorganic nitrogen upon the nitrogen loading provides further support to phosphorus-limitation of most of the US OECD water bodies.

While a positive correlation is seen between nitrogen loading and growing season epilimnetic primary productivity (Figure 45), the data are too scarce to draw any clear conclusions as to the validity of this relationship. It is likely this correlation is

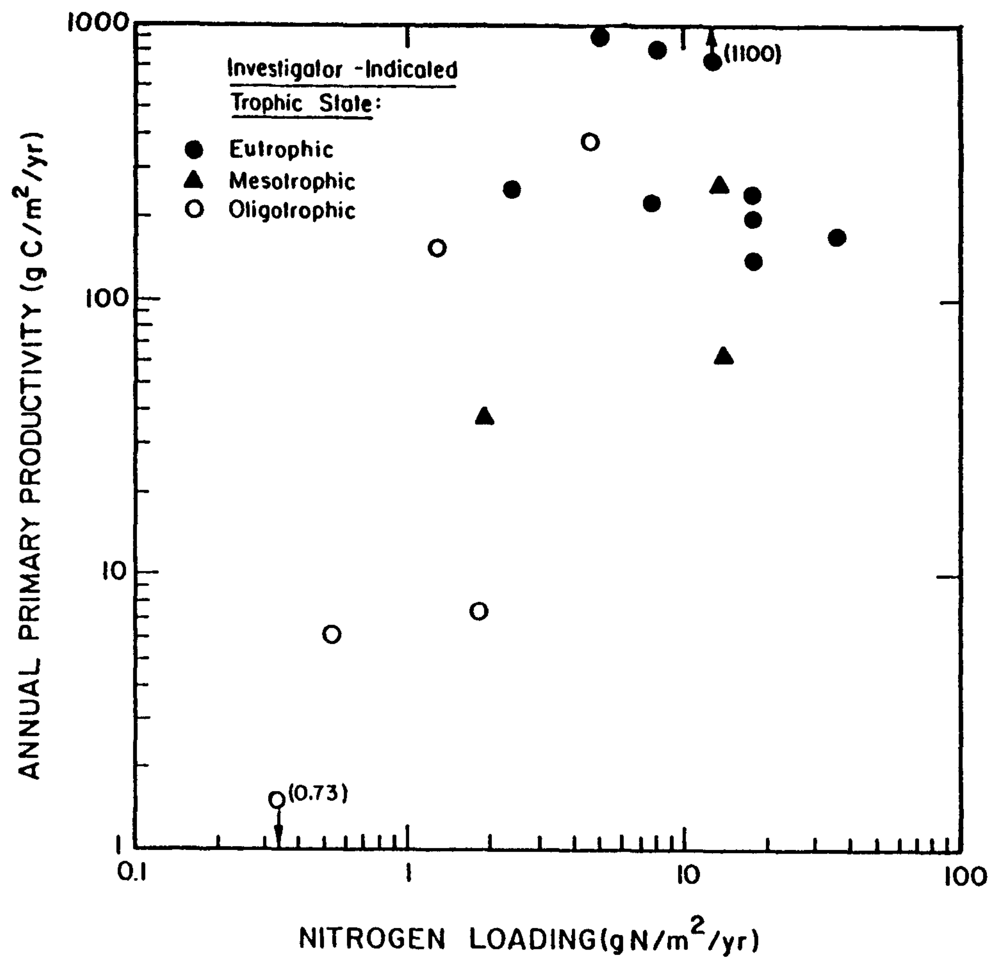

Figure 41. Nitrogen Loading and Primary Productivity Relationship in US OECD Water Bodies

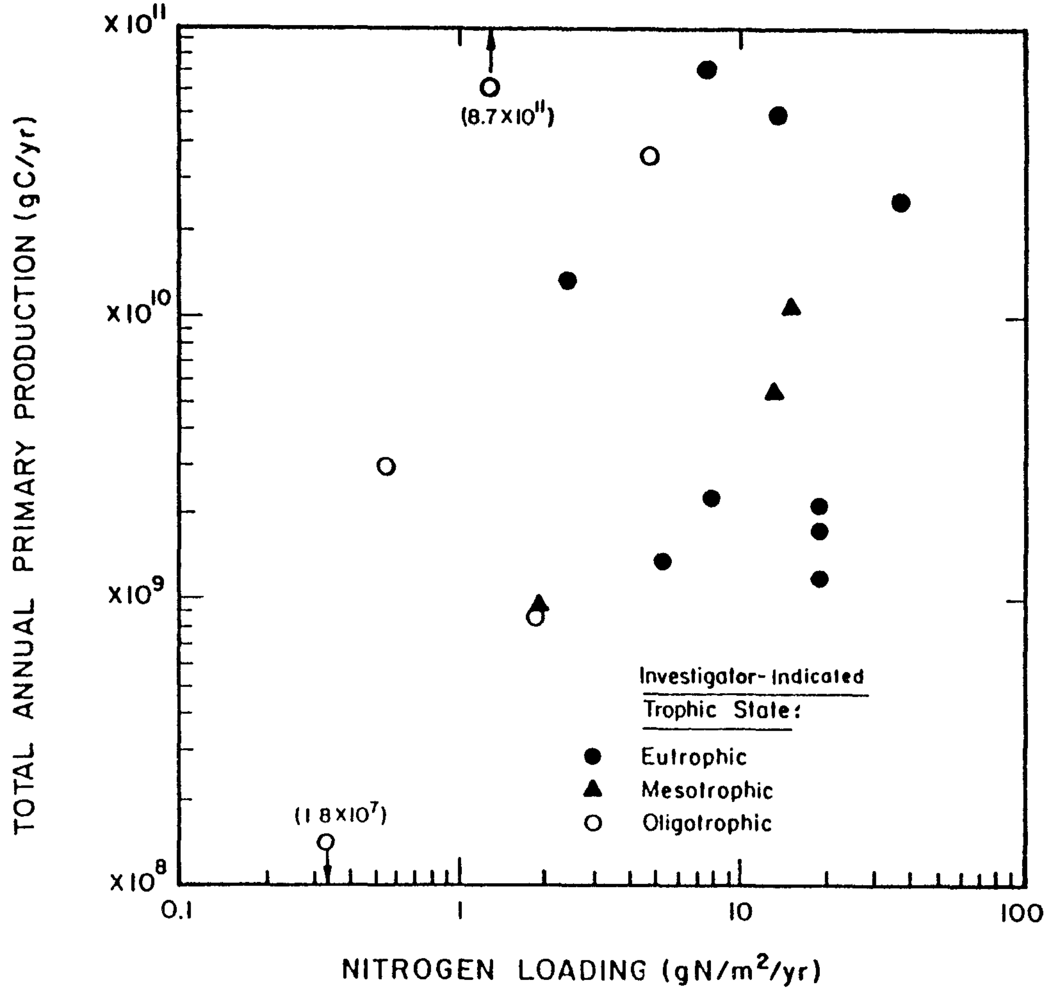

Figure 42. Nitrogen Loading and Total Annual Primary Production Relationship in US OECD Water Bodies

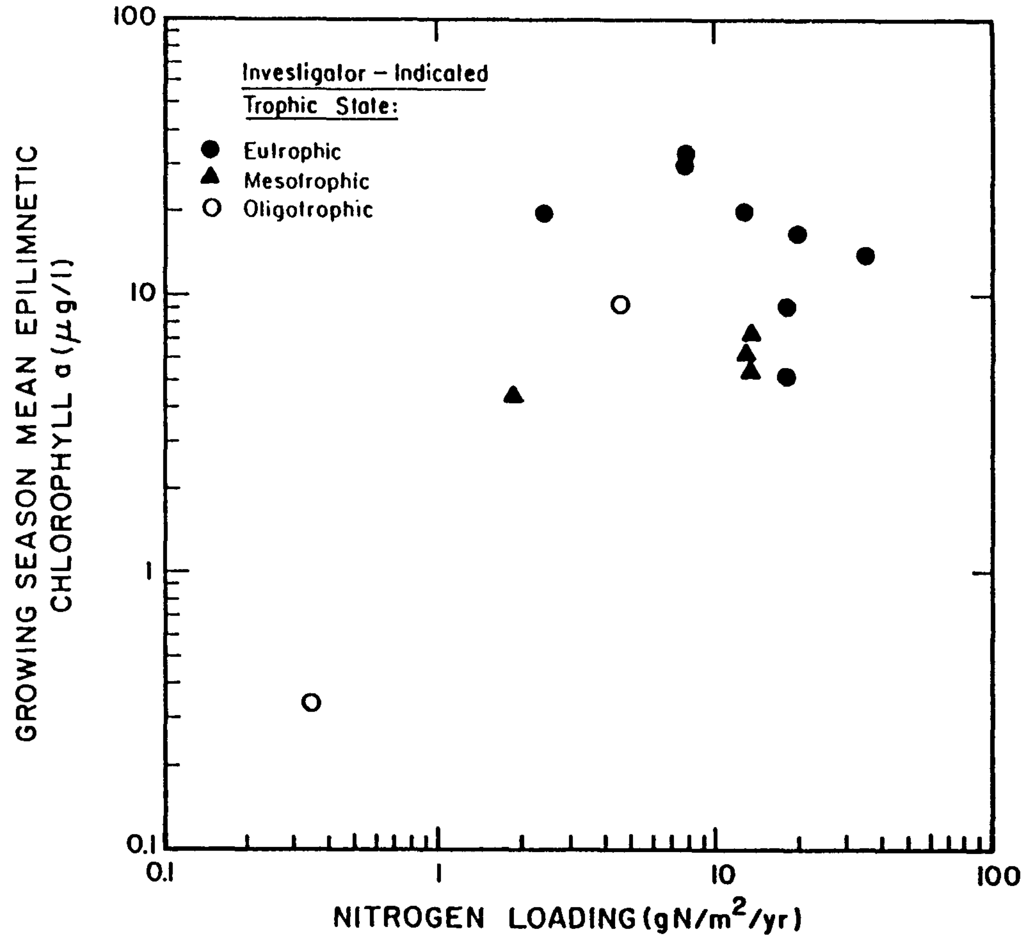

Figure 43. Nitrogen Loading and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

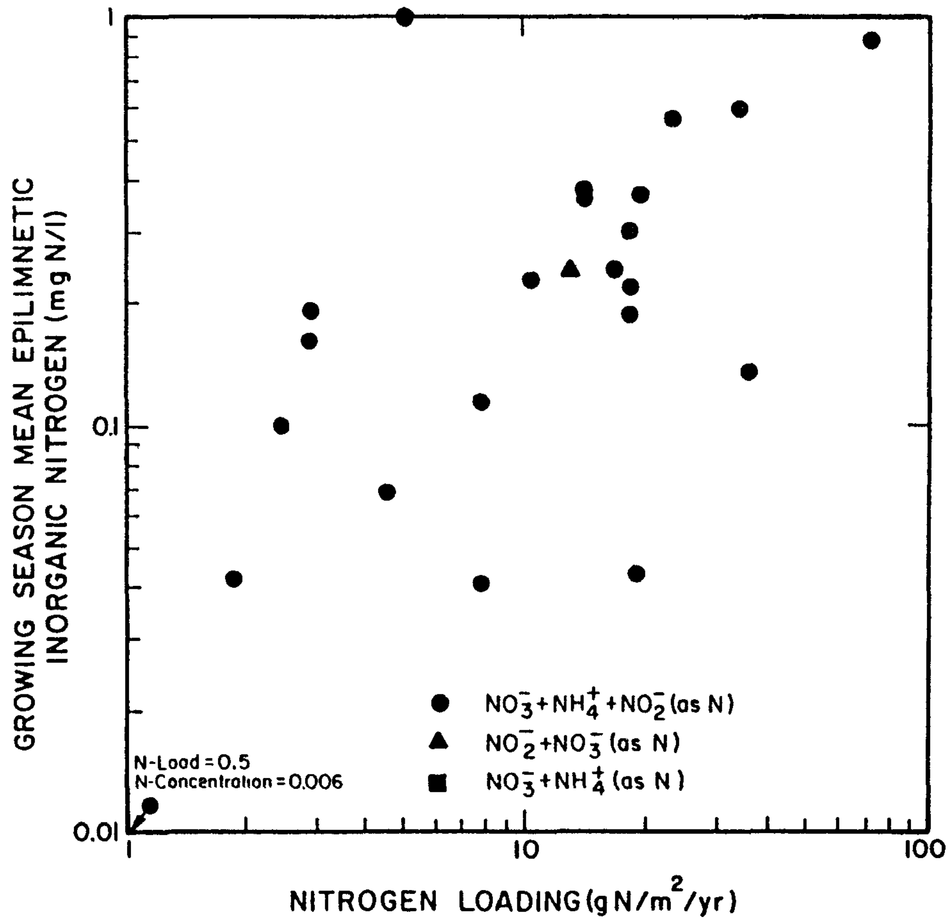

Figure 44. Nitrogen Loading and Growing Season Epilimnetic Inorganic Nitrogen Relationship in US OECD Water Bodies

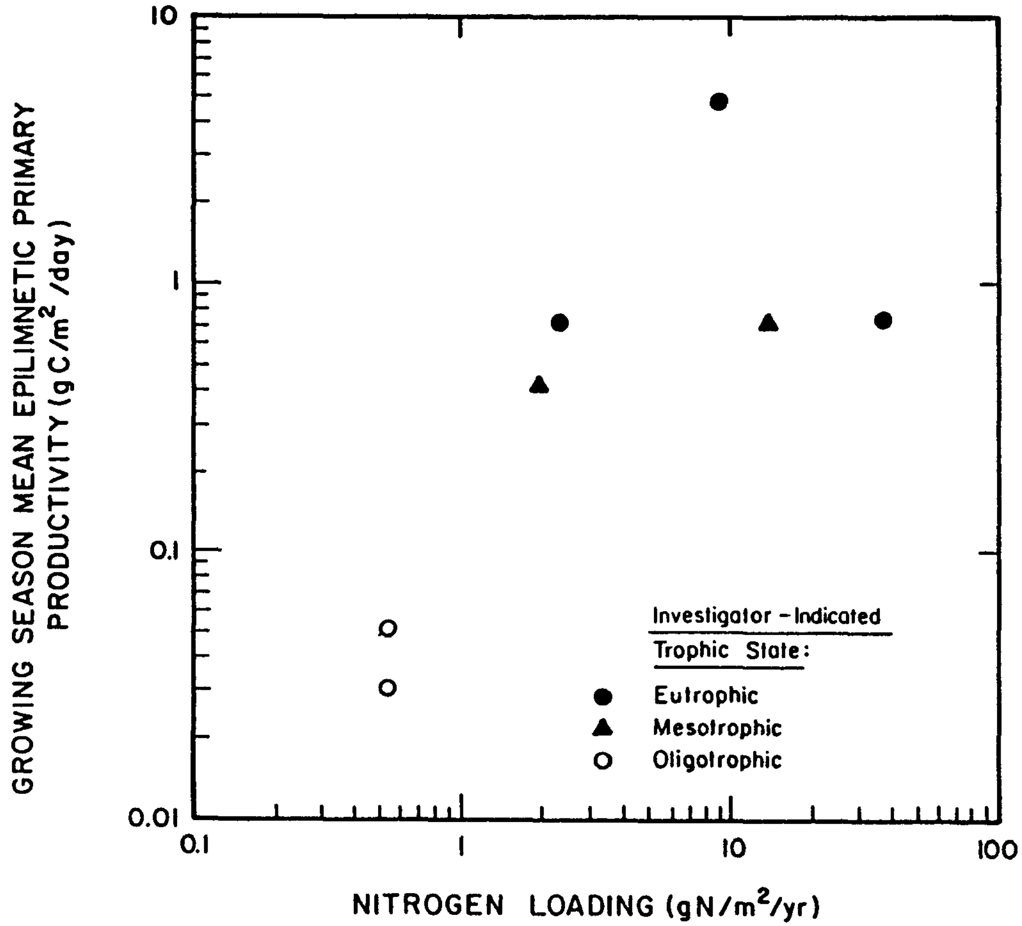

Figure 45. Nitrogen Loading and Growing Season Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

a coincidental artifact of phosphorus-limitation. Finally, there appears to be no correlation between nitrogen loads and the spring overturn concentration of inorganic nitrogen (Figure 46). It should be noted, however, that as with phosphorus (Figure 37), a majority of the data sets include the mean winter concentration rather than the spring overturn concentration. How this difference may affect the results obtained with these correlations is not known.

MEAN TOTAL AND DISSOLVED PHOSPHORUS CONCENTRATIONS

A positive correlation was observed between total phosphorus and mean chlorophyll a in the US OECD water bodies (Figure 47) even though the 'mean' values consisted of annual means, ice-free period means and summer means. Dillon and Rigler (1975) and Jones and Bachmann (1976) have also reported high correlations between these two parameters. A negative correlation was also seen between mean total phosphorus and mean Secchi depth (Figure 48). This is to be expected since Secchi depth is a negative hyperbolic function of the chlorophyll content of a water body (Edmondson, 1972; Carlson, 1974; Dillon and Rigler, 1975). Since chlorophyll is correlated with mean total phosphorus, mean Secchi depth should also be correlated with mean total phosphorus, as was observed. A high positive correlation was noted for the mean total phosphorus and the mean dissolved phosphorus (Figure 49). This is not surprising since the dissolved phosphorus content of the water body should be related to the total phosphorus content. This correlation indicates the dissolved phosphorus appears to be a relatively constant fraction on an annual basis of the total phosphorus in the US OECD water bodies. The mean total phosphorus also appears to be positively correlated with the mean primary productivity (Figure 50). The correlation between these two parameters is better than that seen between the phosphorus or nitrogen loading and mean primary productivity (Figures 31 and 32, respectively).

In general, although positive correlations are noted, the data are too scarce to make any valid conclusions about the relationship between mean total phosphorus and either the growing season epilimnetic chlorophyll a or primary productivity (Figures 51 and 52, respectively). A positive correlation may exist between mean total phosphorus and the spring overturn total phosphorus (Figure 53), although the data are also relatively scarce for this relationship. It should also be noted that a majority of the water bodies in Figure 53 have mean winter total phosphorus concentrations plotted rather than the spring overturn concentrations. It is not known how this affects the results of this correlation, although the effects would likely be small. A positive correlation was also noted between the growing season epilimnetic total phosphorus concentration and the growing season epilimnetic chlorophyll a (Figure 54). While there are more data sets for this correlation than for the relationship between annual mean total phosphorus and growing season chlorophyll a (Figure 51), there is also more scatter of the data. The correlation between the growing season epilimnetic total phosphorus

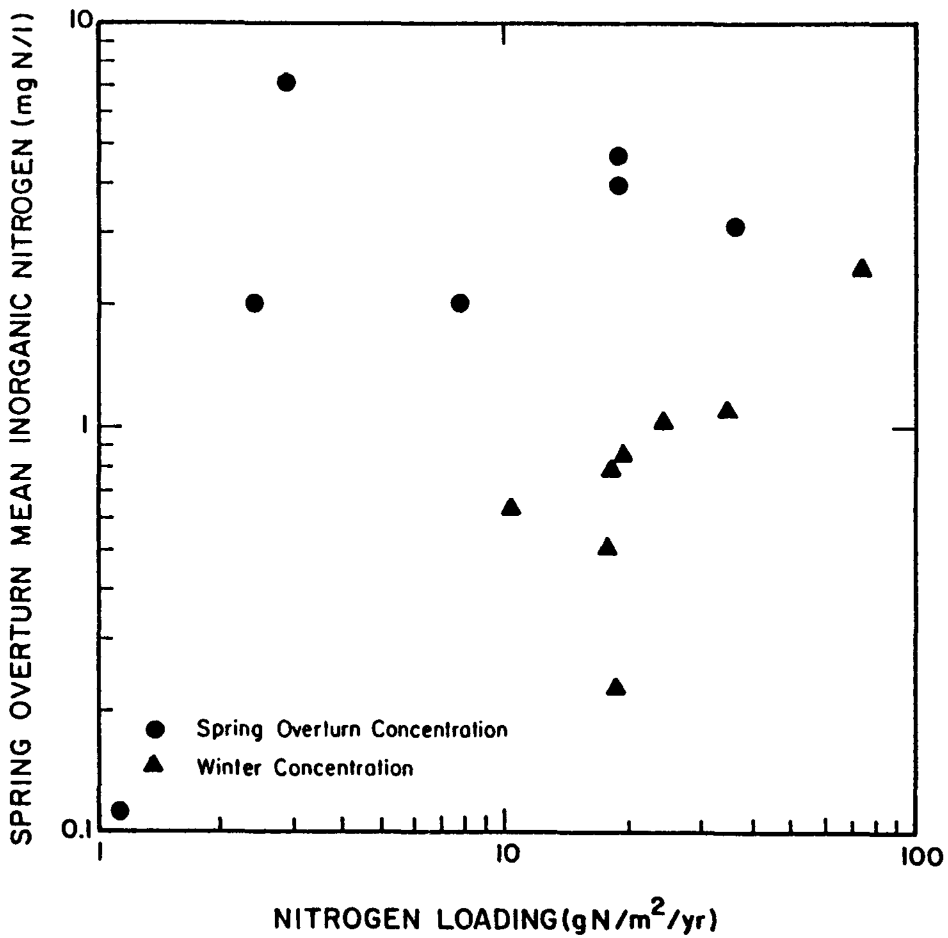

Figure 46. Nitrogen Loading and Spring Overturn Inorganic Nitrogen Relationship in US OECD Water Bodies

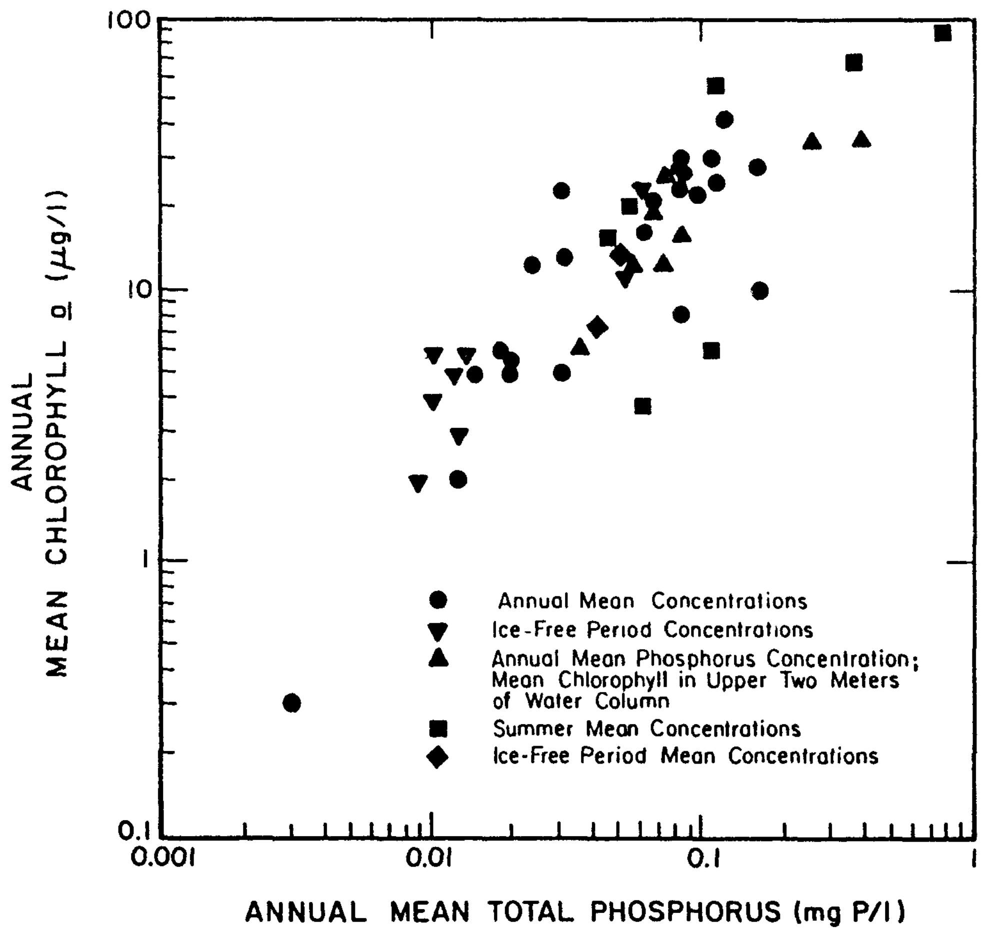

Figure 47. Mean Total Phosphorus and Mean Chlorophyll a Relationship in US OECD Water Bodies

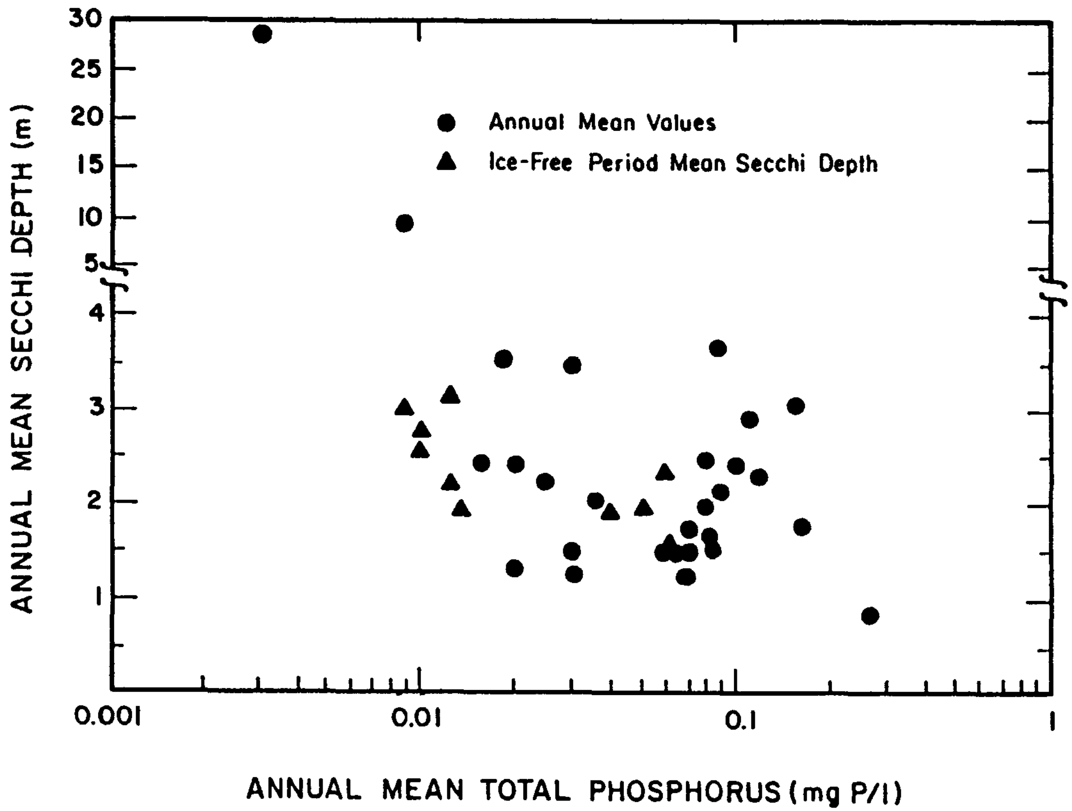

Figure 48. Mean Total Phosphorus and Mean Secchi Depth Relationship in US OECD Water Bodies

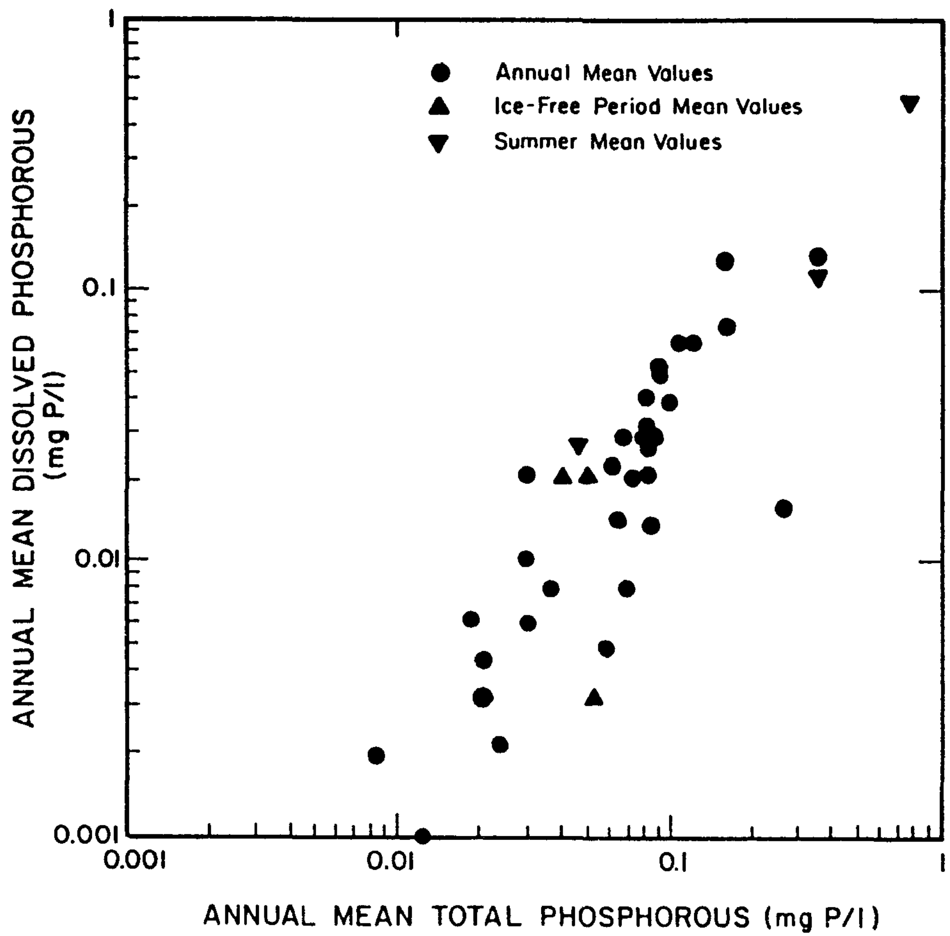

Figure 49. Mean Total Phosphorus and Mean Dissolved Phosphorus Relationship in US OECD Water Bodies

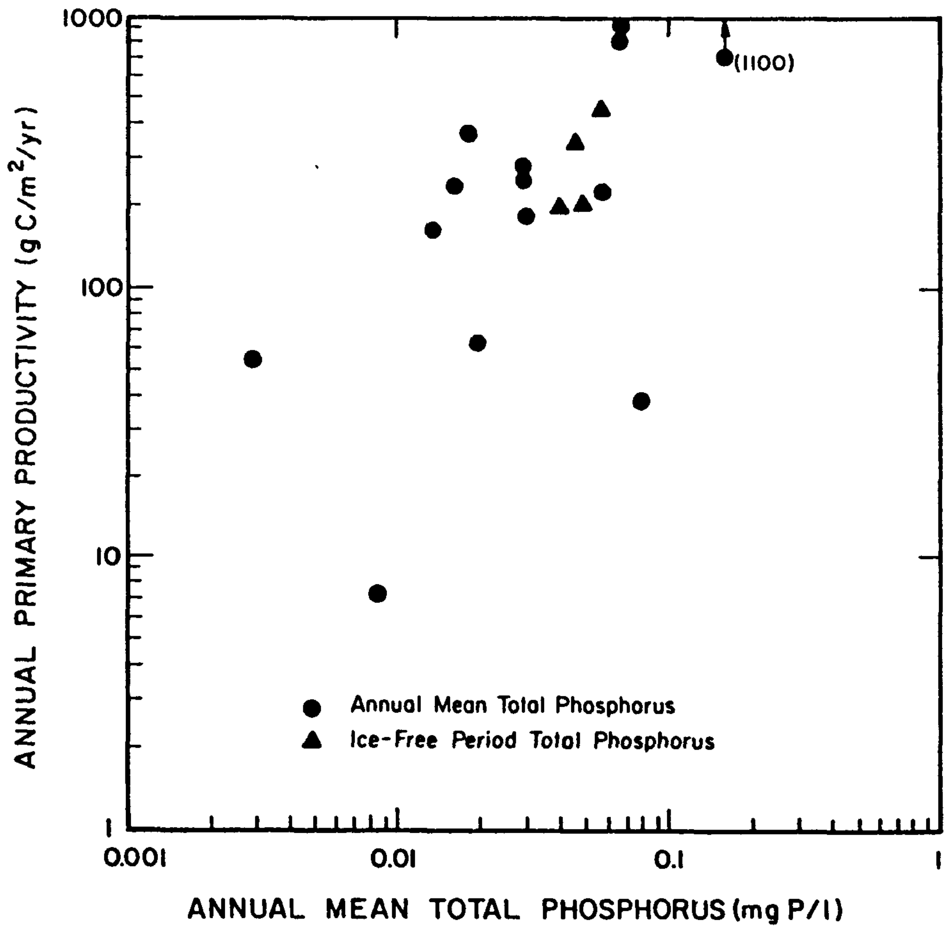

Figure 50. Mean Total Phosphorus and Primary Productivity Relationship in US OECD Water Bodies

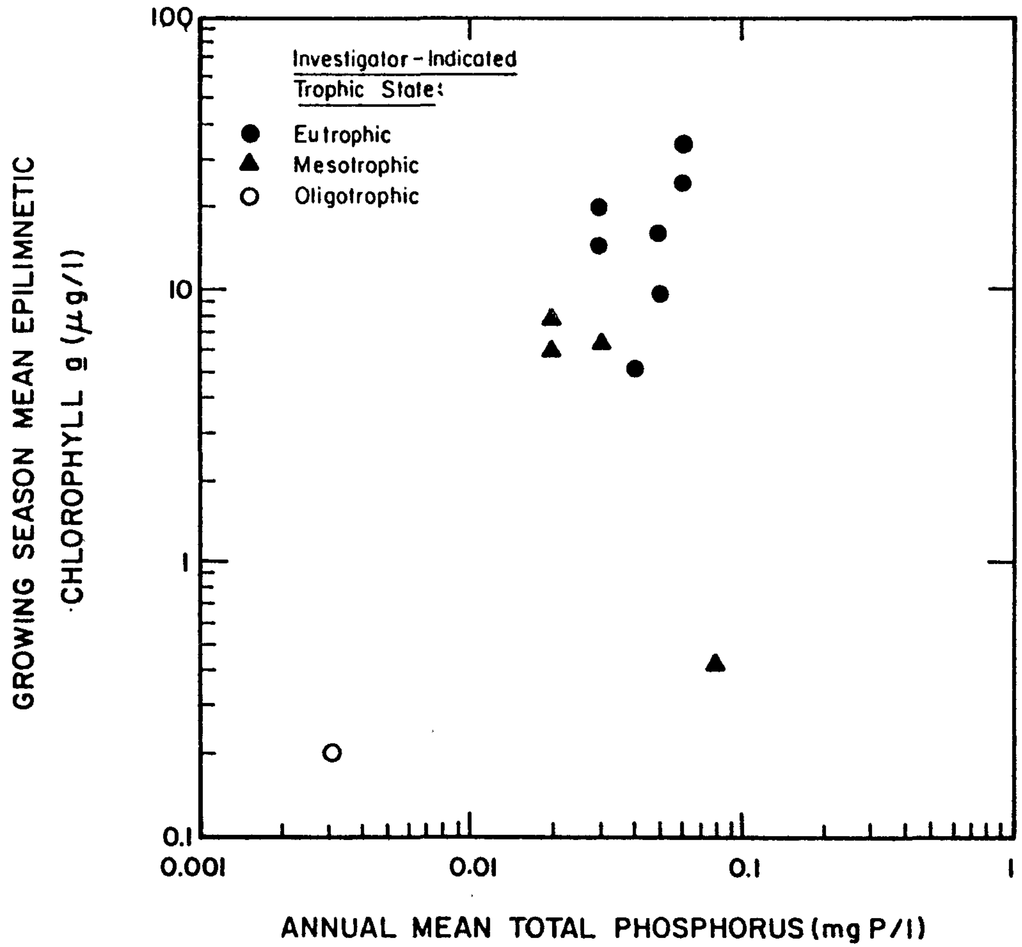

Figure 51. Mean Total Phosphorus and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

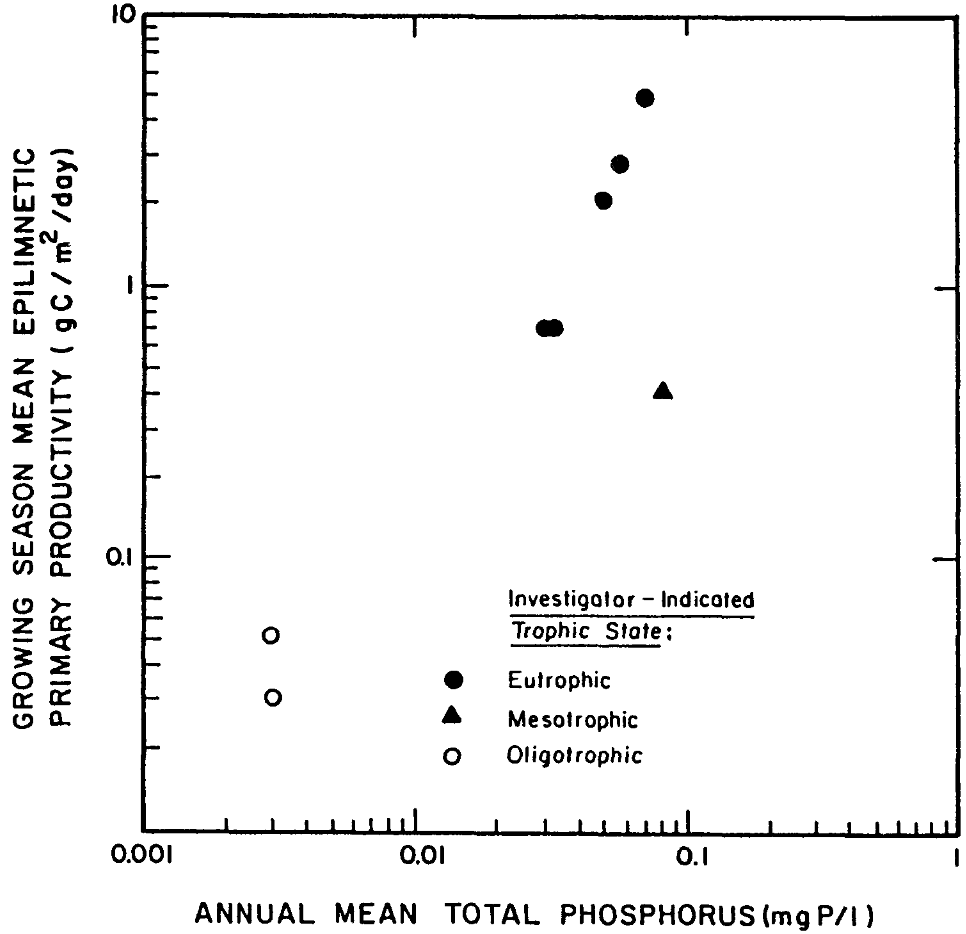

Figure 52. Mean Total Phosphorus and Growing Season Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

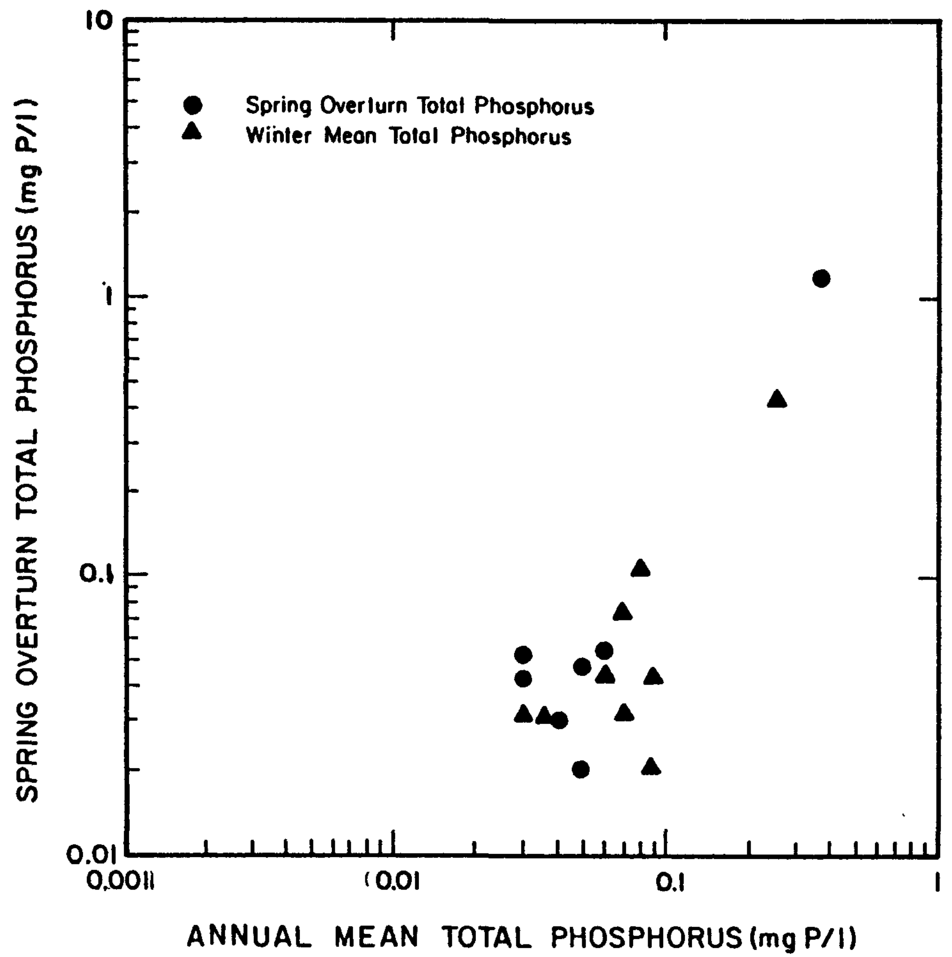

Figure 53. Mean Total Phosphorus and Spring Overturn Total Phosphorus Relationship in US OECD Water Bodies

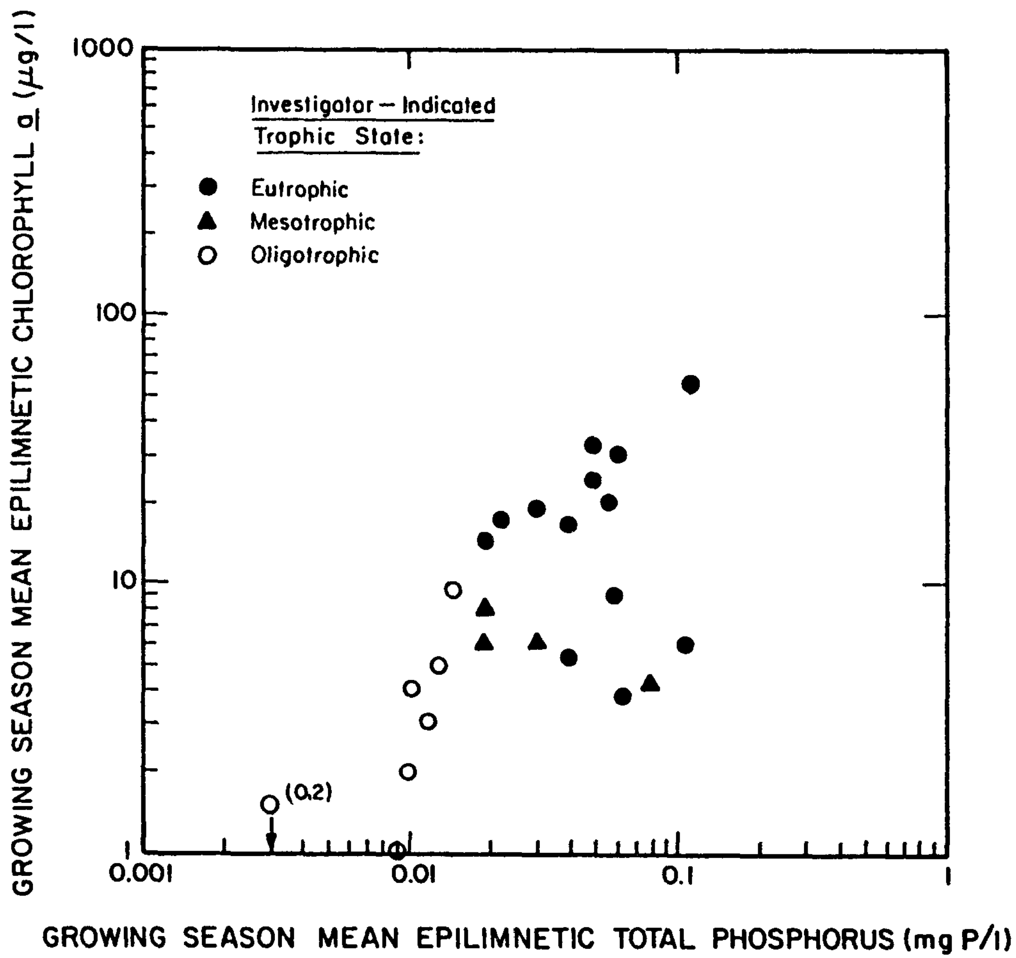

Figure 54. Growing Season Epilimnetic Total Phosphorus and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

and growing season primary productivity (Figure 55) is very similar to that seen with the annual mean total phosphorus (Figure 52). This suggests that the total phosphorus concentration does change significantly over the annual cycle. However, the scarcity of data does not allow for a rigorous examination of these two relationships. A positive correlation was also noted between the spring overturn total phosphorus concentration and the growing season epilimnetic concentrations of chlorophyll a, total phosphorus and dissolved phosphorus (Figures 56, 57 and 58, respectively). However, none of these three relationships had sufficient data for a valid assessment of their degree of correlation. It would particularly have been informative to examine the correlation between the spring overturn total phosphorus concentration and the growing season chlorophyll concentration (Figure 56) since Sakamoto (1966), Dillon and Rigler (1974a) and Vollenweider (1976a) have shown good correlations between these two parameters.

The correlation between spring overturn phosphorus and growing season epilimnetic dissolved phosphorus was also examined (Figure 58). Although there is somewhat of a positive correlation noted, this is not a limnologically logical correlation to consider, since the measured growing season epilimnetic dissolved phosphorus will be the portion of the available phosphorus 'left over' in a water body after the aquatic plant populations have assimilated their metabolic requirements. Consequently, the use of the 'available' nutrients in any of the correlations is of dubious value. They are included in this analysis solely because they were included on the initial list of suggested parameters supplied to all the OECD investigators.

The mean dissolved phosphorus was also included in this eutrophication response parameter analysis but is not expected to yield any useful correlations for the reasons indicated above. There is a possible correlation between the mean dissolved phosphorus and mean chlorophyll a (Figure 59). However, the mean chlorophyll a is composed of annual mean, ice-free period mean and surface mean values. Consequently, little validity was given to this relationship. By contrast, the correlation between the mean total phosphorus and mean chlorophyll a (Figure 47) is much better than that seen for mean dissolved phosphorus. The correlation between mean dissolved phosphorus and primary productivity (Figure 60) partially supports this view. There is considerable scatter in the data for these two parameters which indicates little correlation between them. The primary productivity data is too scarce for correlation, but it is not expected that a larger US OECD data set would show a positive correlation.

There appears to be a positive correlation between mean dissolved phosphorus and spring overturn dissolved phosphorus (Figure 61). This is not unexpected if the dissolved phosphorus is the 'leftover' fraction. Presumably the larger the leftover dissolved phosphorus content of the water body, the larger will be the concentration at spring overturn. However, this correlation shows more data scatter than that between mean total phos-

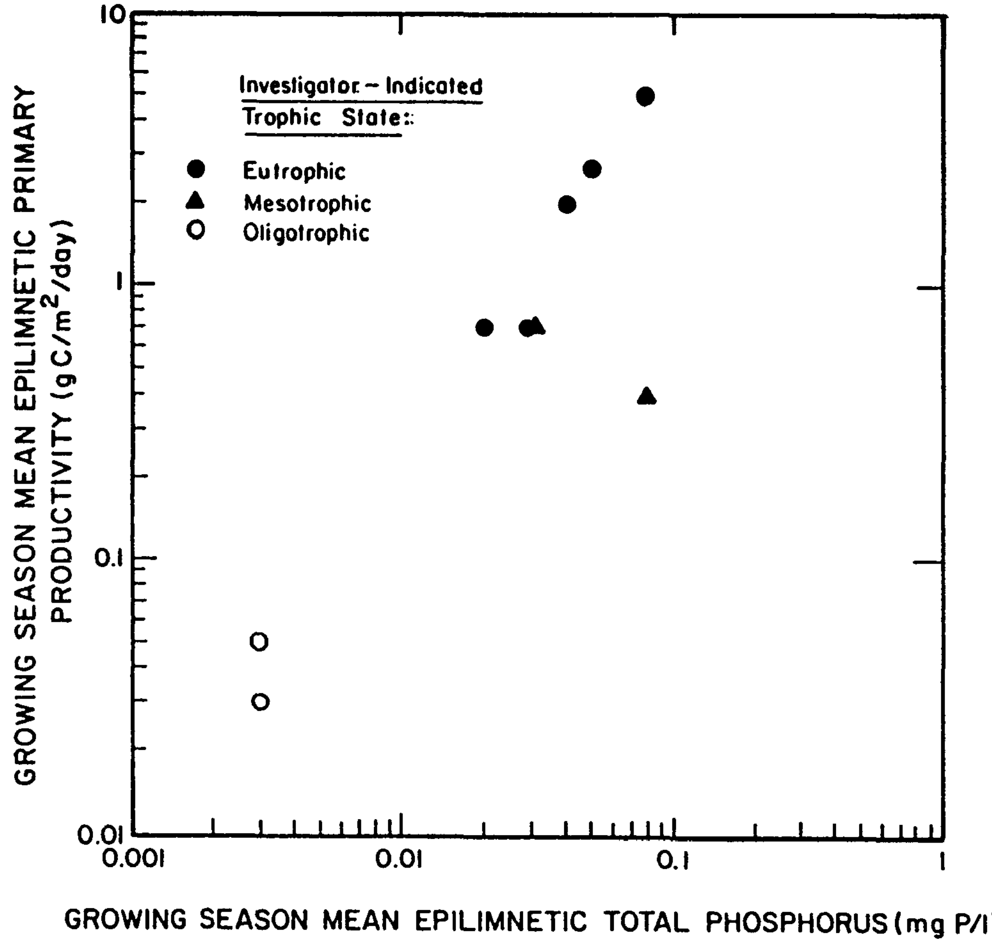

Figure 55. Growing Season Epilimnetic Total Phosphorus and Growing Season Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

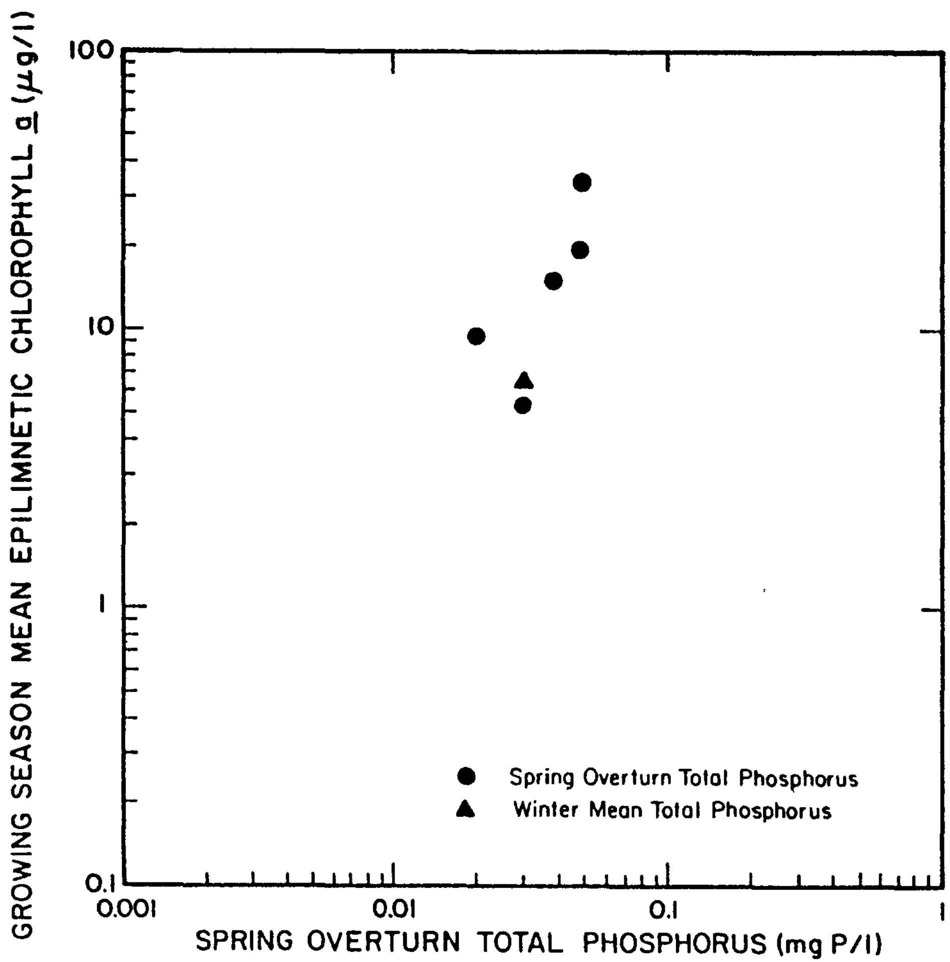

Figure 56. Spring Overturn Total Phosphorus and Growing Season Epilimnetic Chlorophyll *a* Relationship in US OECD Water Bodies

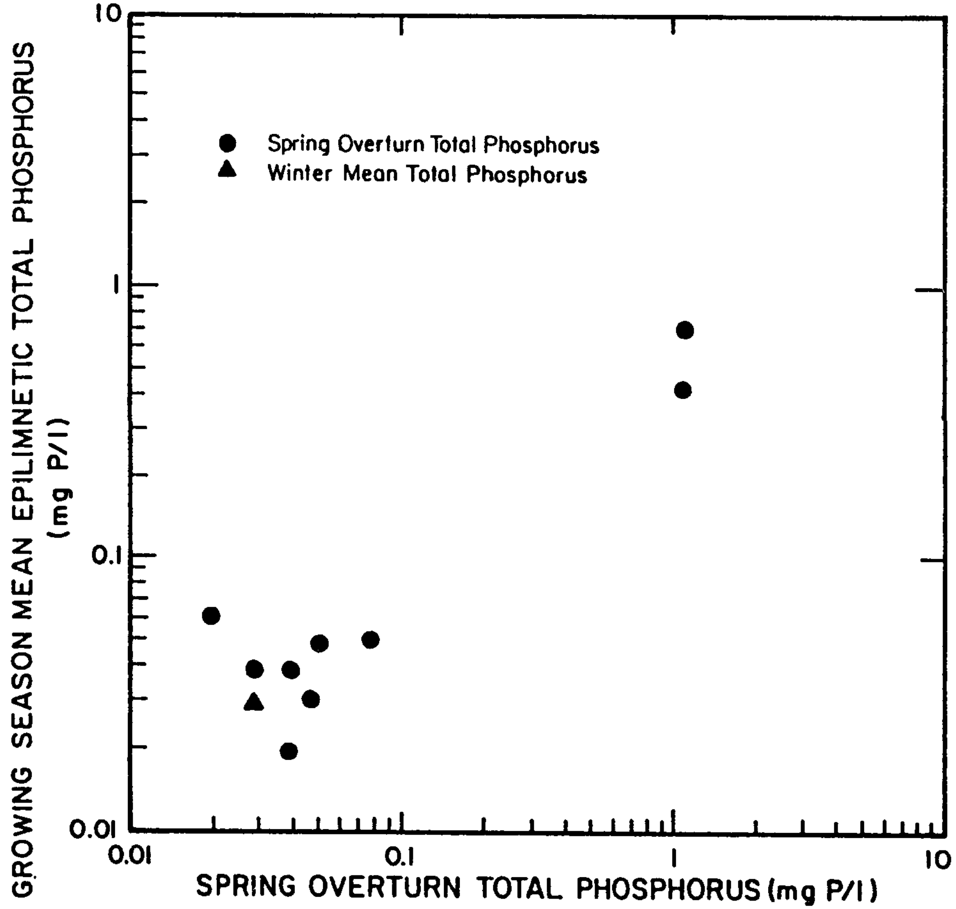

Figure 57. Spring Overturn Total Phosphorus and Growing Season Epilimnetic Total Phosphorus Relationship in US OECD Water Bodies

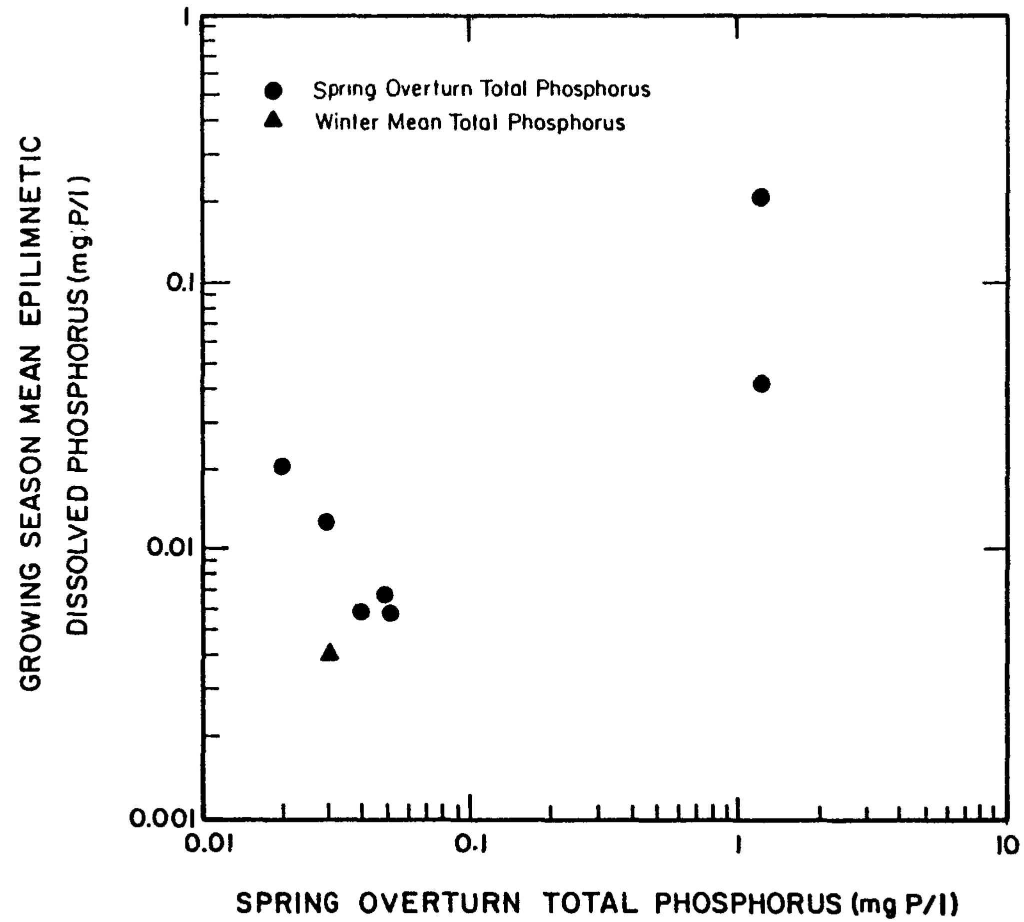

Figure 58. Spring Overturn Total Phosphorus and Growing Season Mean Epilimnetic Dissolved Phosphorus Relationship in US OECD Water Bodies

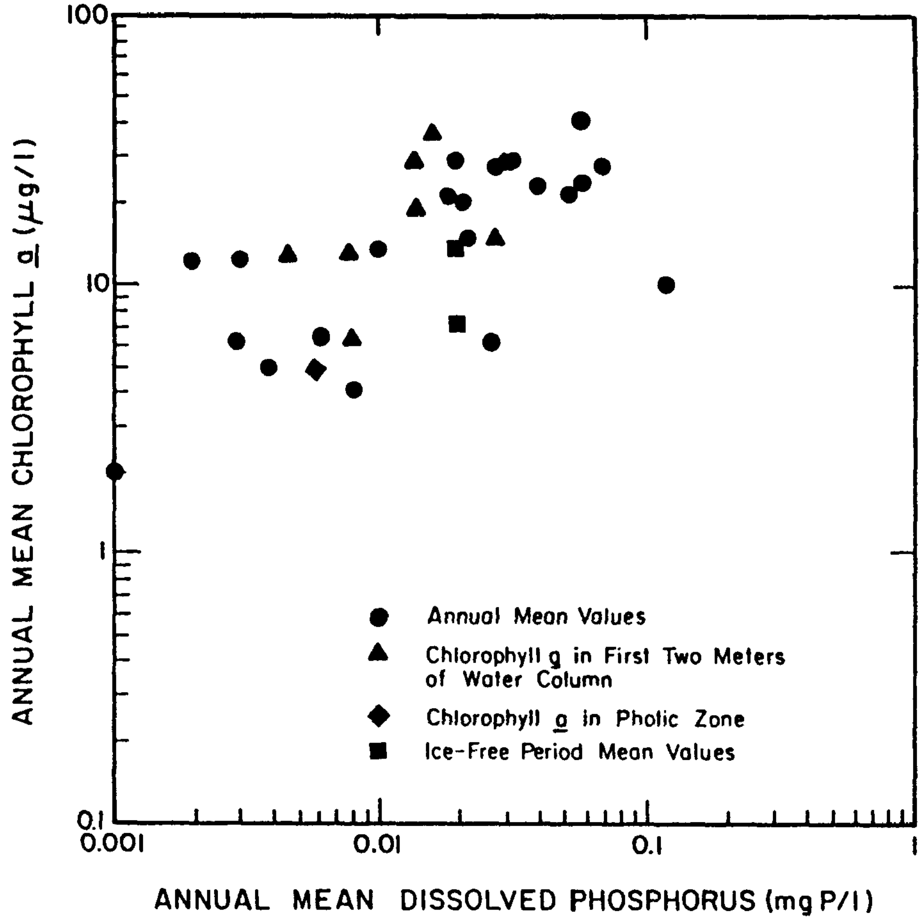

Figure 59. Mean Dissolved Phosphorus and Mean Chlorophyll a Relationship in US OECD Water Bodies

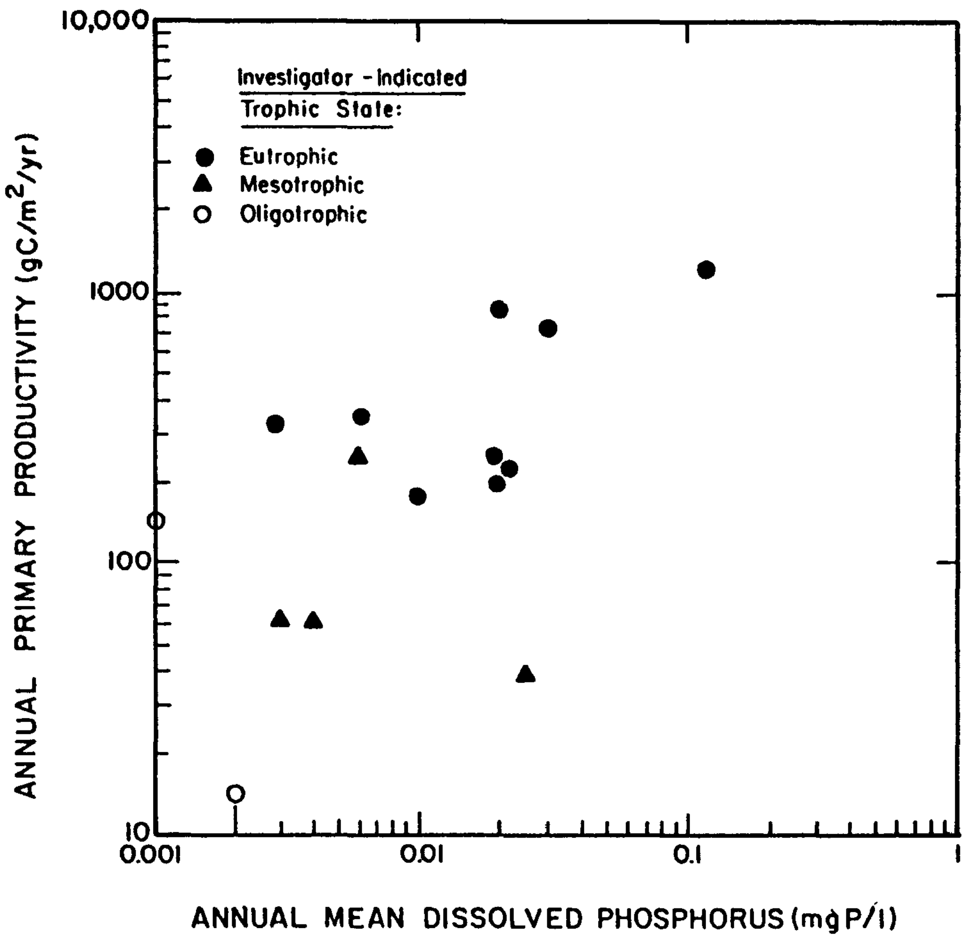

Figure 60. Mean Dissolved Phosphorus and Primary Productivity Relationship in US OECD Water Bodies

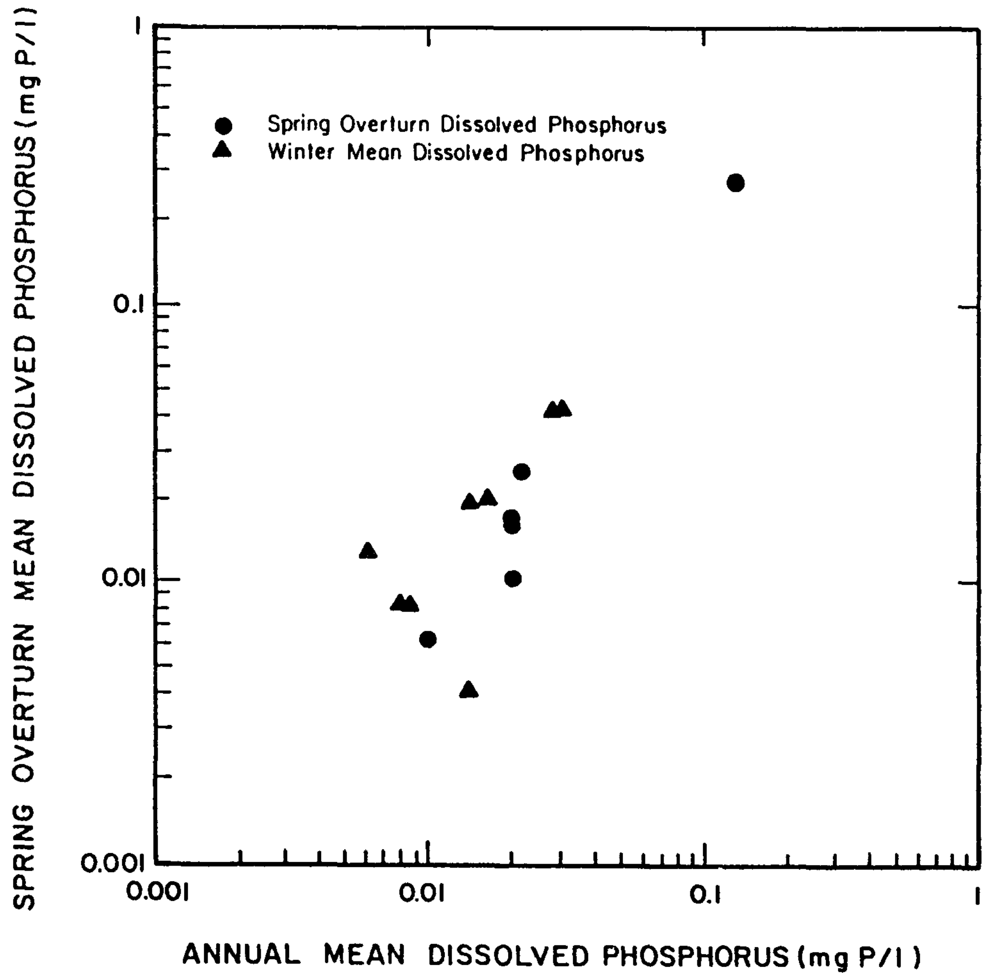

Figure 61. Mean Dissolved Phosphorus and Spring Overturn Dissolved Phosphorus Relationship in US OECD Water Bodies

phorus and spring overturn total phosphorus, as would be expected. There also appears to be no readily observable correlation between either the growing season epilimnetic dissolved phosphorus or the spring overturn dissolved phosphorus and the growing season epilimnetic chlorophyll a (Figures 62 and 63, respectively). However, the data set for these two correlations is very small, which precludes a rigorous analysis of these relationships. It is possible a positive correlation may exist between the spring overturn dissolved phosphorus concentration and the growing season epilimnetic chlorophyll a in phosphorus-limited water bodies. A positive correlation is noted between the spring overturn dissolved phosphorus and the growing season epilimnetic dissolved phosphorus (Figure 64). However, in addition to a data set which is too small for a valid evaluation of this relationship, a positive correlation between these two parameters is not limnologically consistent with the conditions normally found in phosphorus-limited water bodies. The available US OECD data sets for these two parameters are almost completely for phosphorus-limited waters. Consequently, the apparent correlation is probably an artifact. There were not sufficient data to examine the correlation between spring overturn dissolved phosphorus and growing season epilimnetic primary productivity. Presumably, if it existed, the correlation would be a positive one.

MEAN INORGANIC NITROGEN CONCENTRATIONS

Before examination of correlations between the mean inorganic (i.e., algal-available) nitrogen and eutrophication response parameters, it should be noted that the concentrations of this algal nutrient, as with dissolved phosphorus, will rise and fall as a function of the algal activity in a water body. Thus, as before, this nitrogen fraction will represent the 'leftover' nitrogen after the algal populations have assimilated their stochiometric requirements for growth. Hence, an observed correlation may be an artifact of this process. It is further complicated because the majority of the US OECD water bodies are phosphorus-limited. Therefore, the leftover inorganic nitrogen concentrations will likely always be higher than the available dissolved phosphorus concentration. The same inorganic nitrogen forms were not reported for all US OECD water bodies. Some investigators reported the mean concentration of $NH_4^+ + NO_3^- + NO_2^-$ (as N) and others reported $NH_4^+ + NO_3^-$ (as N), while still others reported $NO_3^- + NO_2^-$ (as N). These various combinations were treated as equal components in the correlations, although it is not correct to do so. These factors should be considered when examining any correlations between inorganic nitrogen and other eutrophication response parameters in the US OECD water bodies.

There appears to be little correlation between mean inorganic nitrogen and mean chlorophyll a (Figure 65). Removal of the one outlying point at low annual mean inorganic nitrogen and chlorophyll a results in a situation in which there is essentially no relationship between the two parameters. By contrast, there is a

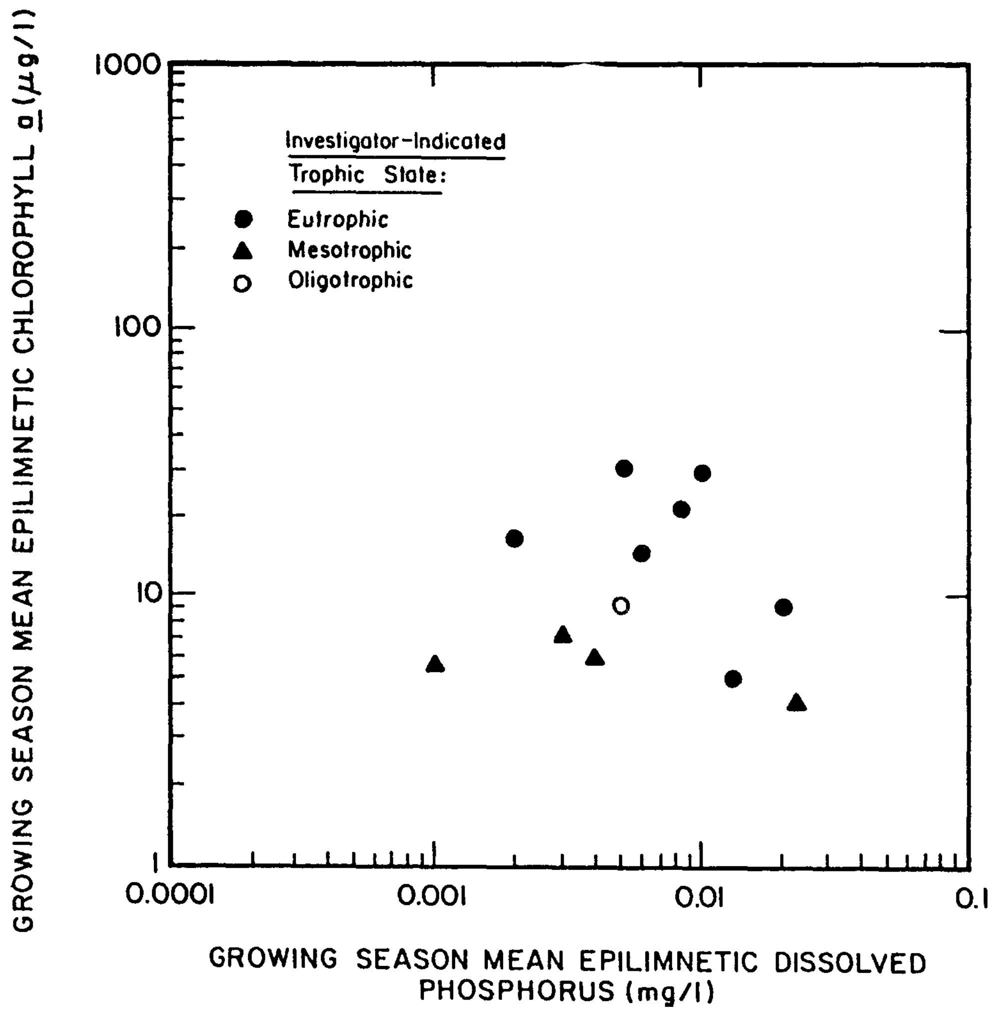

Figure 62. Growing Season Epilimnetic Dissolved Phosphorus and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

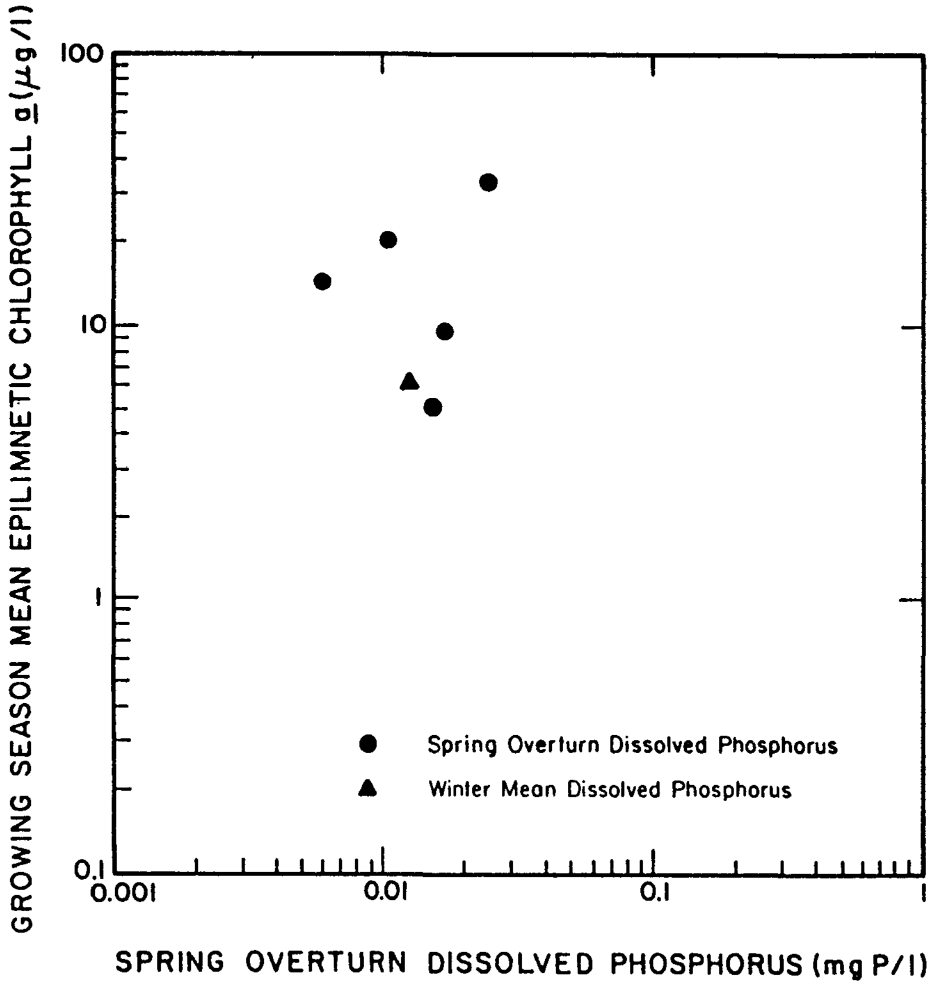

Figure 63. Spring Overturn Dissolved Phosphorus and Growing Season Mean Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

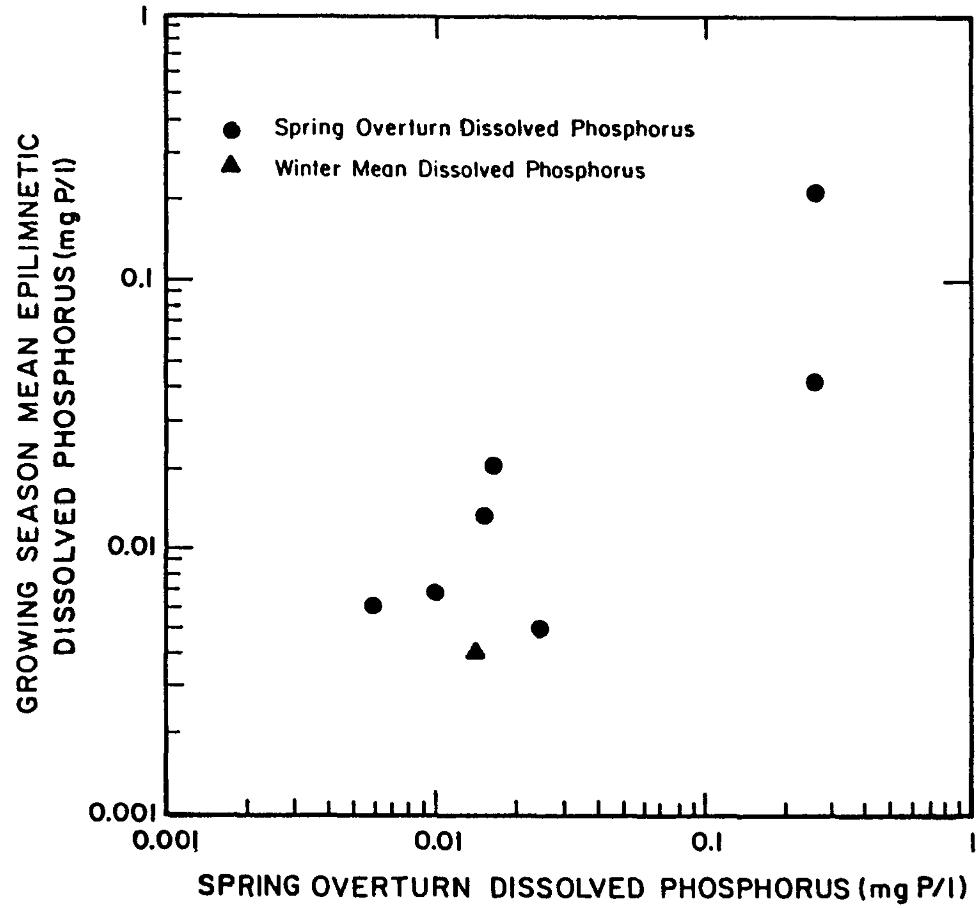

Figure 64. Spring Overturn Dissolved Phosphorus and Growing Season Epilimnetic Dissolved Phosphorus Relationship in US OECD Water Bodies

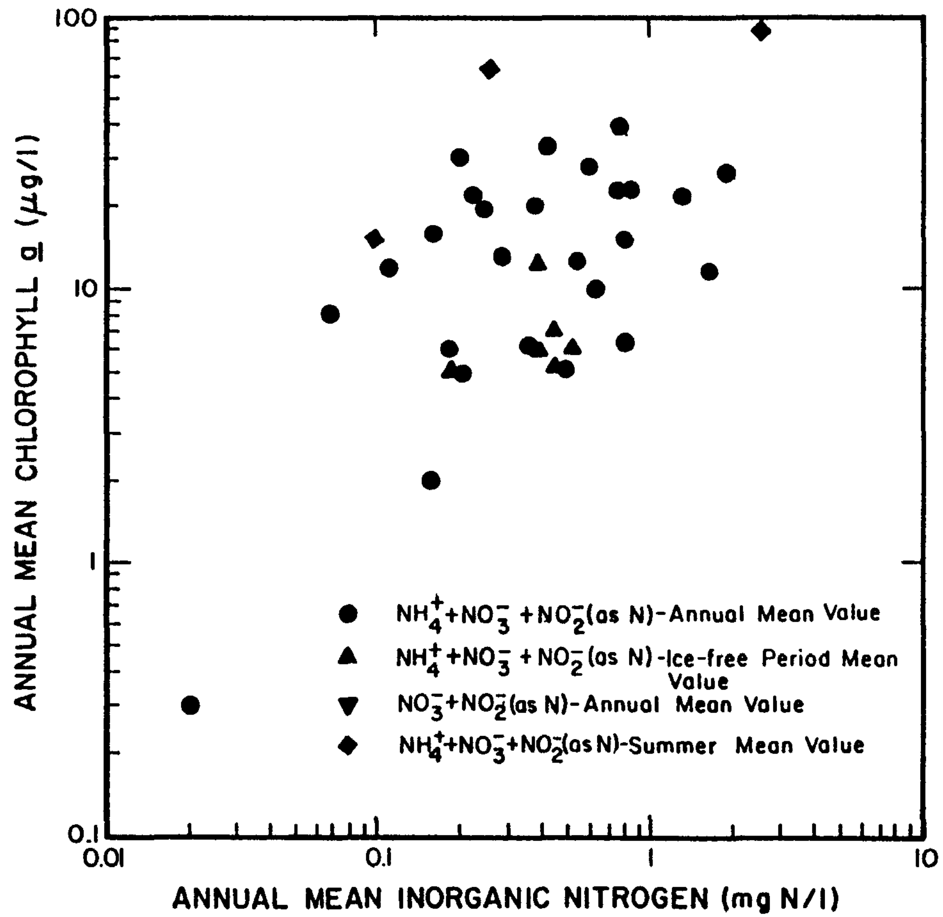

Figure 65. Mean Inorganic Nitrogen and Mean Chlorophyll a Relationship in US OECD Water Bodies

good correlation between mean total phosphorus and mean chlorophyll a (Figure 47). This observation further substantiates the importance of phosphorus, rather than nitrogen, in controlling algal growth. There is little or no correlation between mean inorganic nitrogen and mean Secchi depth (Figure 66). There is a positive correlation between mean inorganic nitrogen and primary productivity (Figure 67). This further supports the phosphorus-limitation of most US OECD water bodies. If the water bodies were nitrogen-limited, one would expect a negative correlation between these two parameters. In fact, the opposite correlation is indicated in Figure 67. By contrast, the poor correlation between the dissolved phosphorus and the primary productivity (Figure 60) illustrates its controlling role in the eutrophication process in the majority of the US OECD water bodies.

There is a correlation between mean inorganic nitrogen and the growing season epilimnetic chlorophyll a (Figure 68). A negative correlation would be expected if nitrogen were the controlling algal nutrient. Such a correlation was not seen in Figure 68. A strong positive correlation appears to exist between the mean inorganic nitrogen and the growing season epilimnetic primary productivity (Figure 69). However, there are only about a half dozen data sets for this correlation. This scarcity of data precludes any rigorous evaluation of this correlation. The positive, rather than negative, correlation suggests that nitrogen does not control the algal populations. The lack of data does not allow one to evaluate the correlation between mean inorganic nitrogen and spring overturn inorganic nitrogen.

Interestingly, a negative correlation appears to exist between the growing season epilimnetic inorganic nitrogen and the growing season epilimnetic chlorophyll a (Figure 70). Although the data set is somewhat limited, the correlation appears to be real. This indicates that, while phosphorus may control the algal populations in most of the US OECD water bodies (Figure 62), the need for available nitrogen for algal growth results in a decreased nitrogen concentration during the growing season. A positive correlation also appears to exist between the growing season epilimnetic inorganic nitrogen and the growing season epilimnetic primary productivity (Figure 71). While the data sets are relatively scarce for this correlation, it is consistent with the views expressed above for Figure 70.

OTHER CORRELATIONS BETWEEN EUTROPHICATION RESPONSE PARAMETERS

Several other correlations were also examined in this section, as indicated in Table 26. These latter correlations are grouped together because they are of a varied nature. They are discussed below. There is a positive correlation observed between the mean chlorophyll a and primary productivity (Figure 72) in the US OECD water bodies. Some scatter of the data is observed. One would normally expect a good correlation between these two parameters. This is supported somewhat by the correlation between the primary

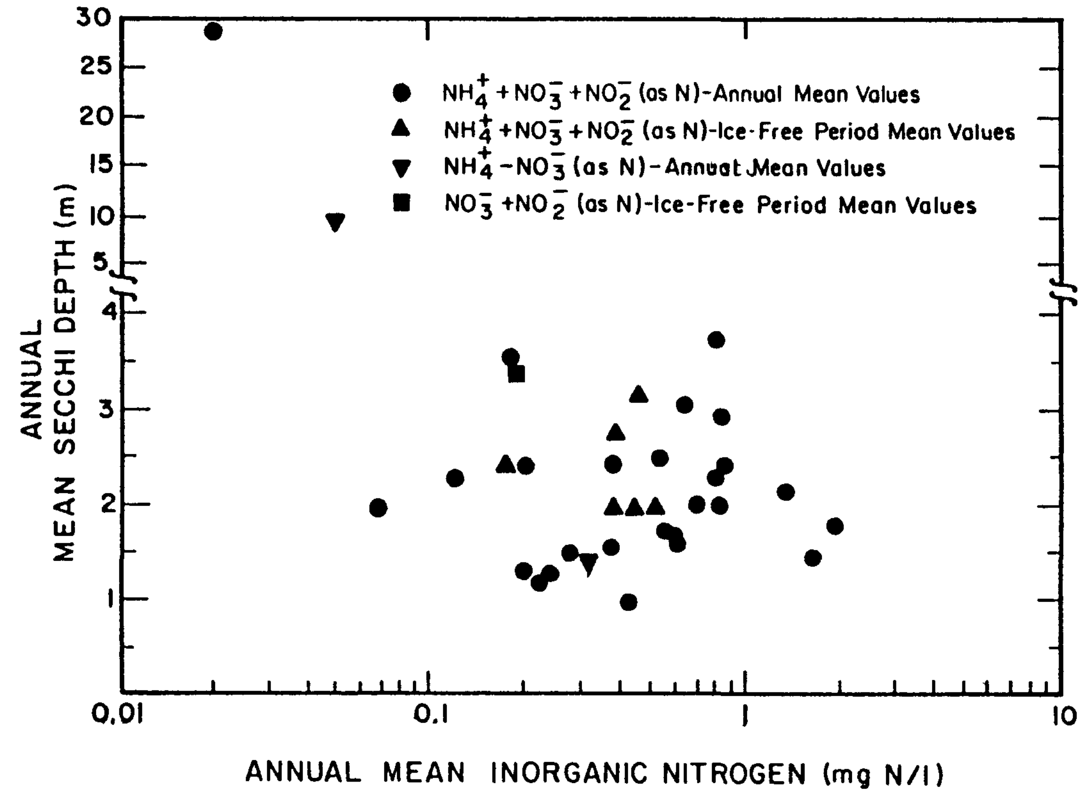

Figure 66. Mean Inorganic Nitrogen and Mean Secchi Depth Relationship in US OECD Water Bodies

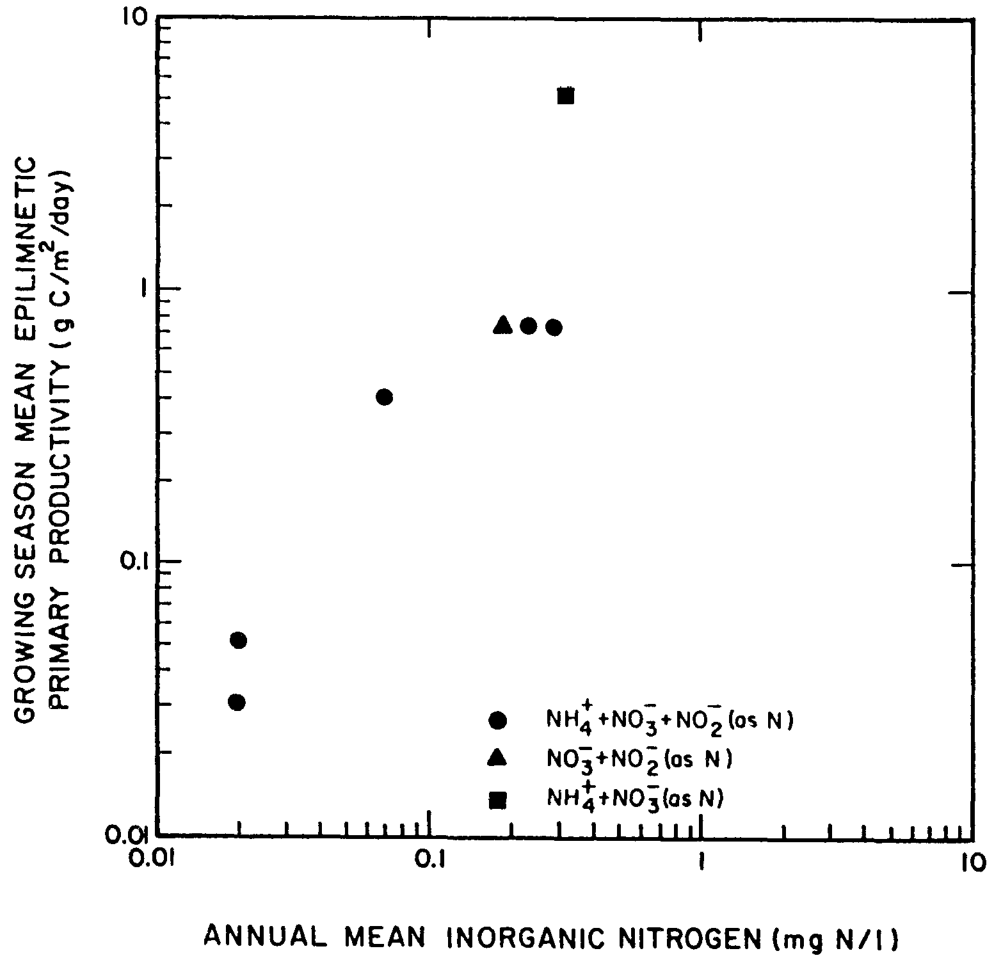

Figure 67. Mean Inorganic Nitrogen and Growing Season Mean Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

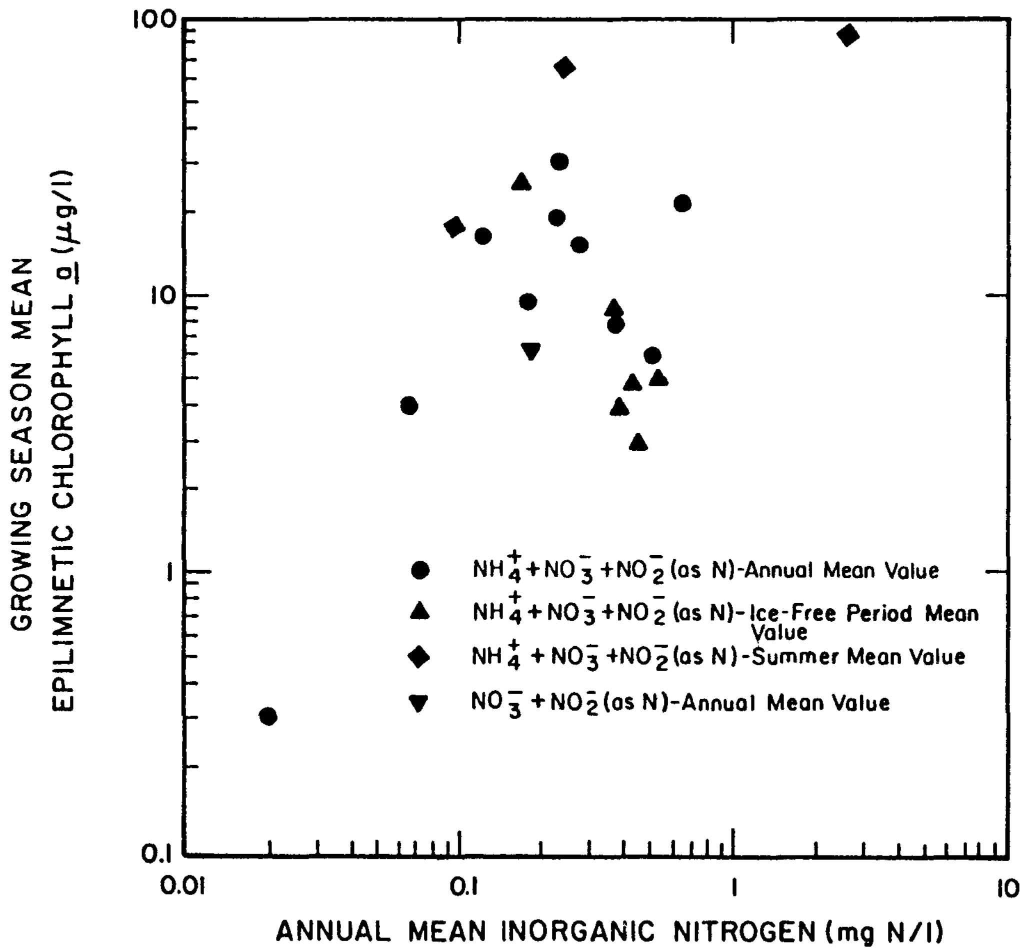

Figure 68. Mean Inorganic Nitrogen and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

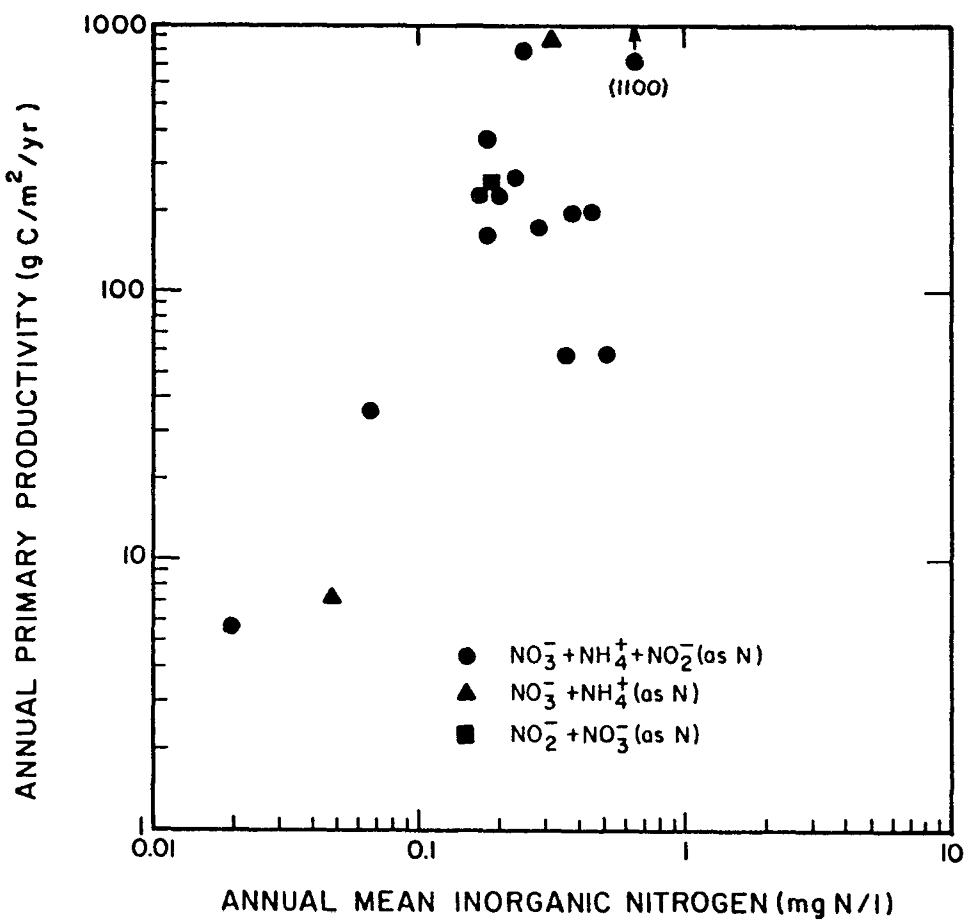

Figure 69. Mean Inorganic Nitrogen and Primary Productivity Relationship in US OECD Water Bodies

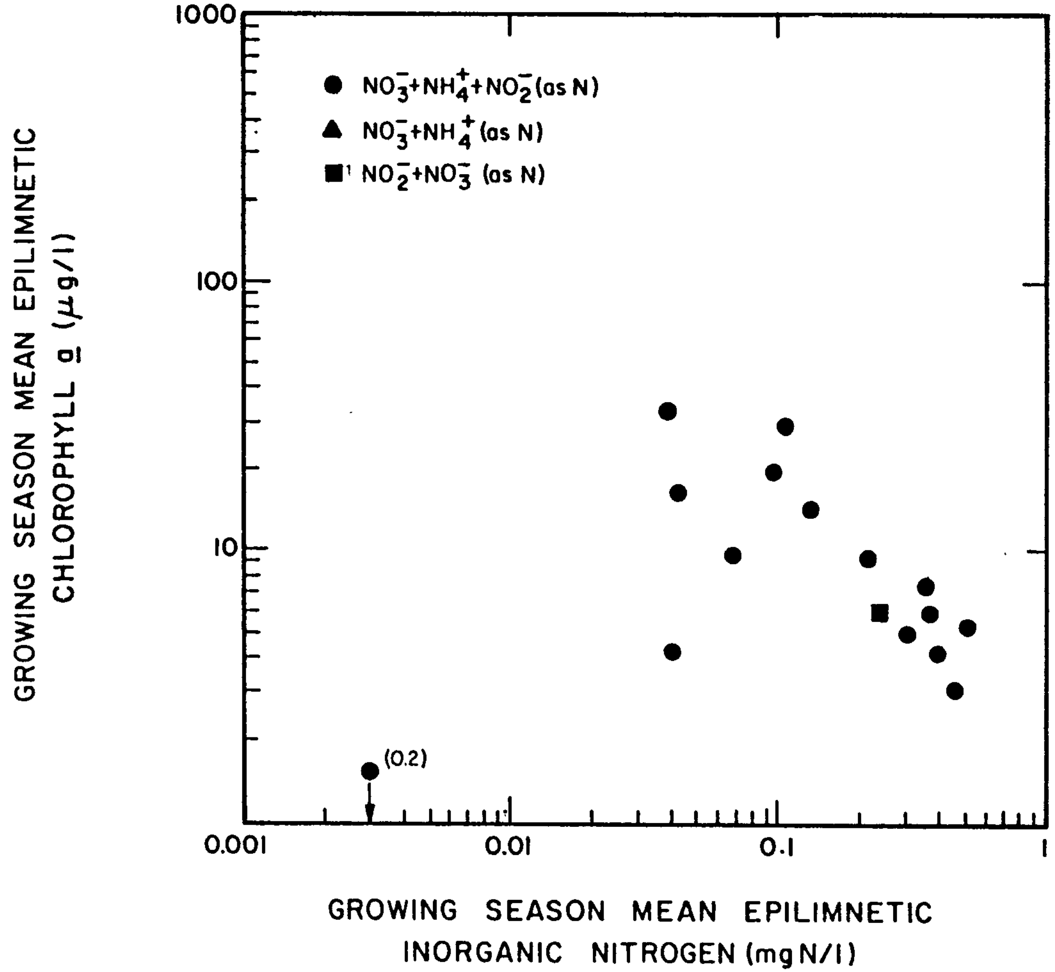

Figure 70. Growing Season Epilimnetic Inorganic Nitrogen and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

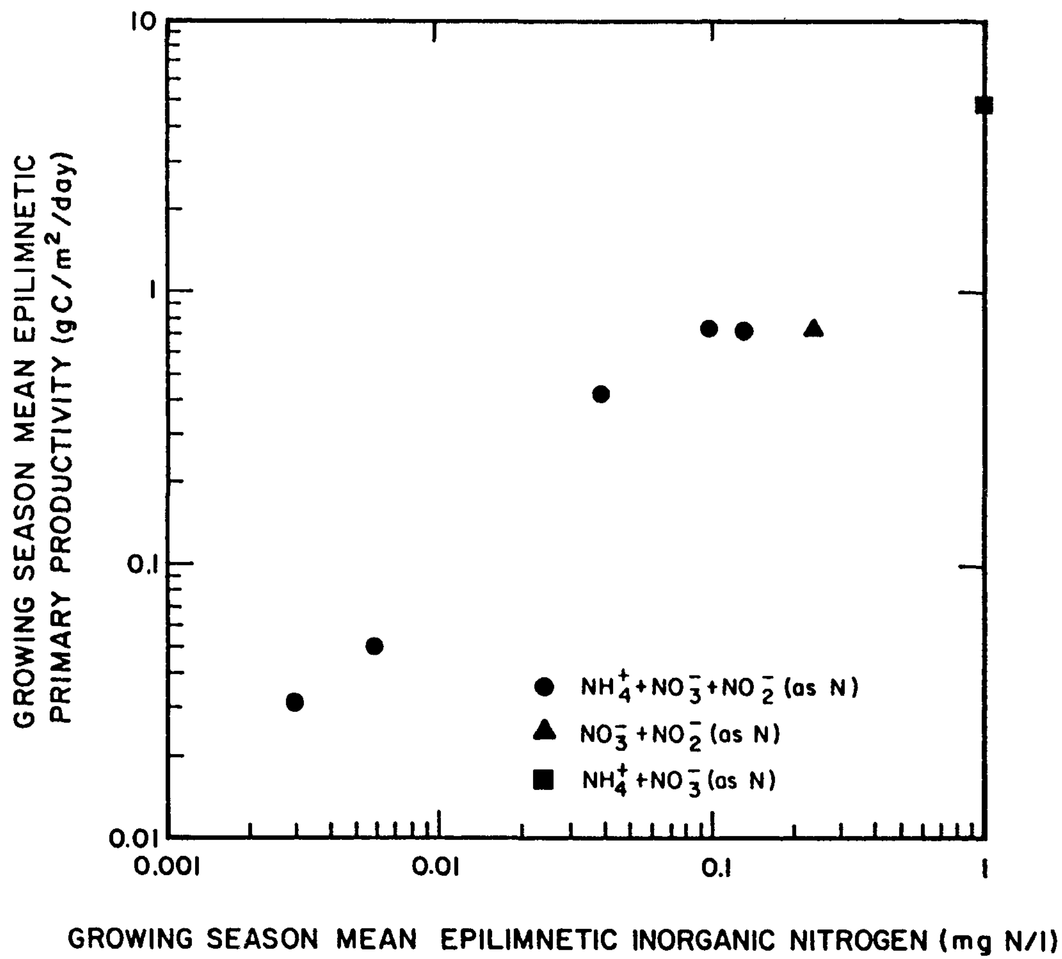

Figure 71. Growing Season Epilimnetic Inorganic Nitrogen and Growing Season Epilimnetic Primary Productivity Relationship in US OECD Water Bodies

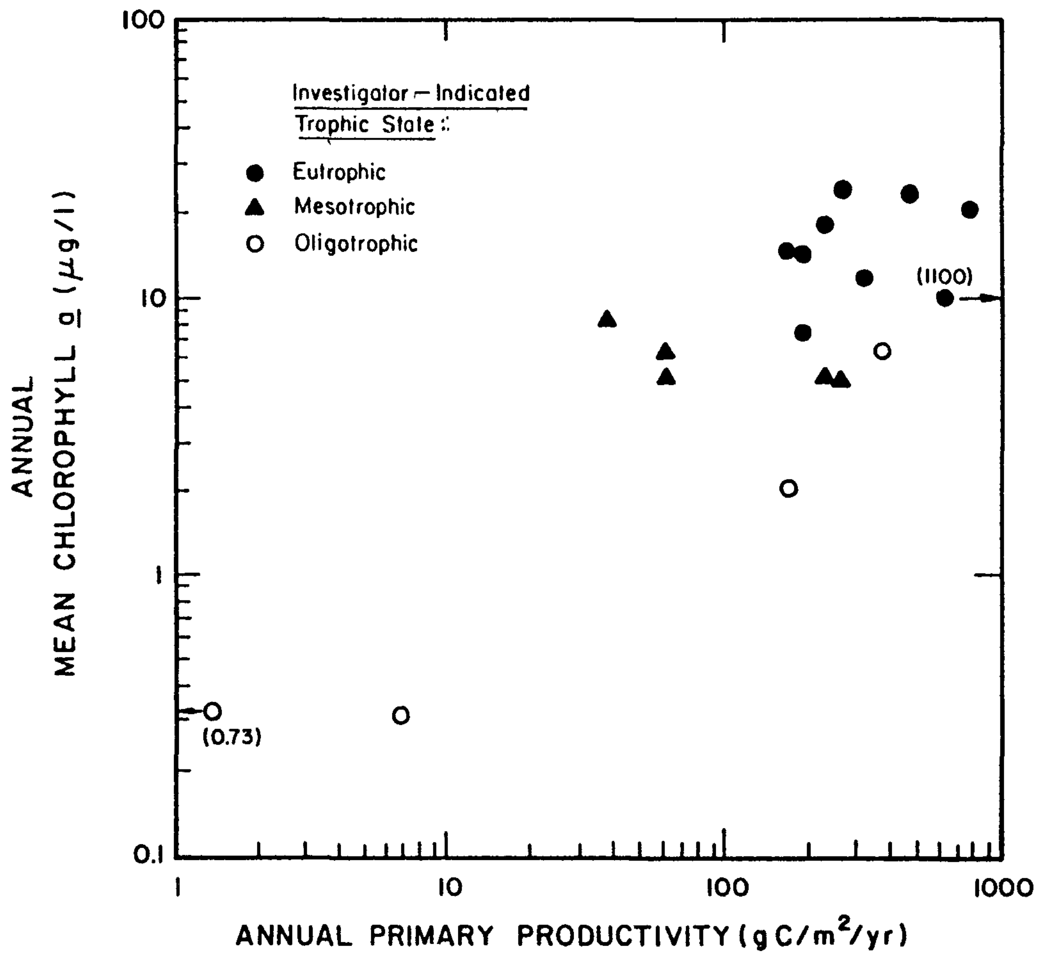

Figure 72. Primary Productivity and Mean Chlorophyll a Relationship in US OECD Water Bodies

productivity and the mean Secchi depth (Figure 73). As expected, there is a negative correlation between these two parameters, although the data are sparse. However, the correlation exhibits a considerable data scatter, which limits its value. It is noted that a majority of the water bodies have primary productivities ranging from about 40-1000 g C/m^2/yr, yet have Secchi depths between 1-3 meters.

A possible hyperbolic relationship was exhibited in the correlation between mean chlorophyll a and mean Secchi depth. The significance of this relationship as a "trigger" for public response to eutrophic water bodies, and as a simple, practical method for measuring water quality was discussed by Edmondson (1972) and Carlson (1974). This correlation is discussed in detail in a later section of this report (see Figures 77 and 78) and serves as the basis of a nutrient load-water quality model developed in this study. There is a positive correlation between the growing season epilimnetic chlorophyll a and primary productivity (Figure 74). This is consistent with the observations indicated above between the annual mean values of these two parameters. There is a somewhat better correlation between the growing season epilimnetic values, as would be expected. However, the very few data sets limit the usefulness of this correlation as a predictive tool.

The annual mean primary productivity, on a daily basis, was correlated with the annual mean chlorophyll a on both a volumetric and areal basis (Figures 75 and 76, respectively). This correlation was analyzed solely because it appeared on the list of suggested correlations. There is a positive correlation between the daily average primary productivity and the annual mean chlorophyll a concentration (Figure 75). This correlation is similar to that observed between the annual mean primary productivity and annual mean chlorophyll a (Figure 72), except that the annual primary productivity is expressed on a daily basis instead of an annual basis. Consequently, Figure 75 yields no more additional information than is already noted in Figure 72. There is little or no correlation between the annual mean daily primary productivity and the annual mean areal chlorophyll a (Figure 76). This data set exhibits a considerable scatter. There appears to be no readily observable advantage in expressing mean chlorophyll a concentrations on an areal basis instead of a volumetric basis.

In conclusion, there appear to be better correlations between the phosphorus loads and concentrations of the US OECD water bodies and the various eutrophication response parameters indicated above than for the nitrogen loads and concentrations. Consistent with phosphorus-limitation of the US OECD water bodies, there are generally poor correlations between the dissolved (i.e., algal-available) phosphorus concentrations and the response parameters examined in the US OECD water bodies. While correlations also existed between the nitrogen loads and concentrations and response parameters of the US OECD water bodies, it is felt that many of these correlations are coincidental artifacts caused by a relatively

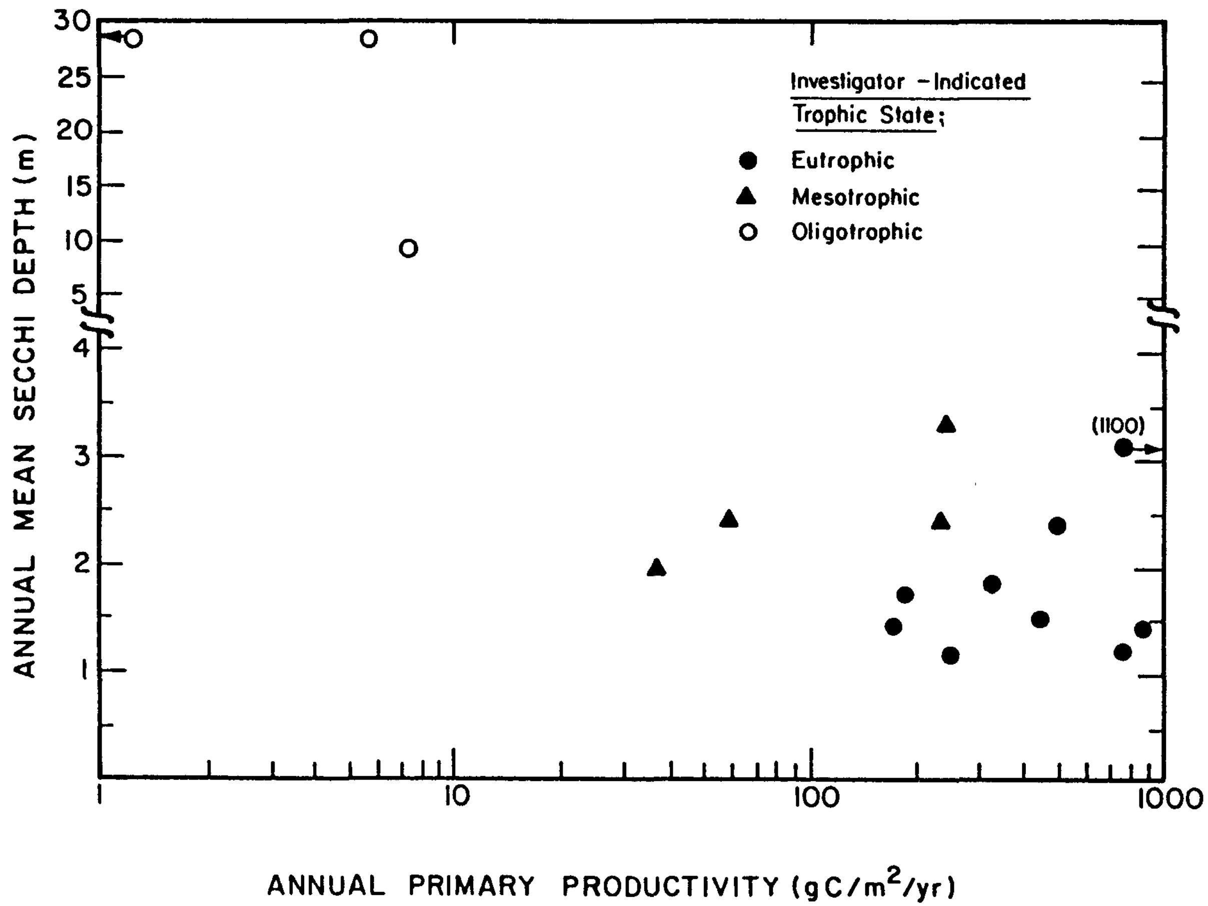

Figure 73. Primary Productivity and Mean Secchi Depth Relationship in US OECD Water Bodies

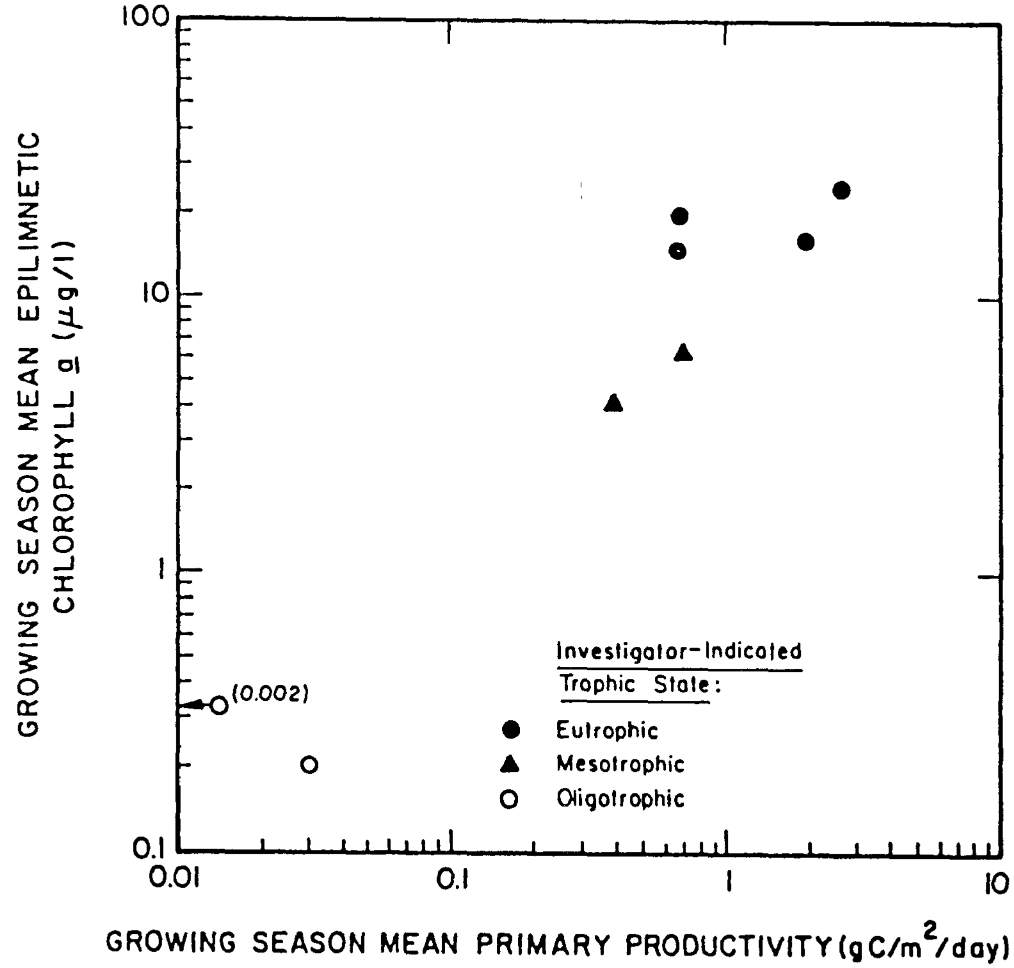

Figure 74. Growing Season Epilimnetic Primary Productivity and Growing Season Epilimnetic Chlorophyll a Relationship in US OECD Water Bodies

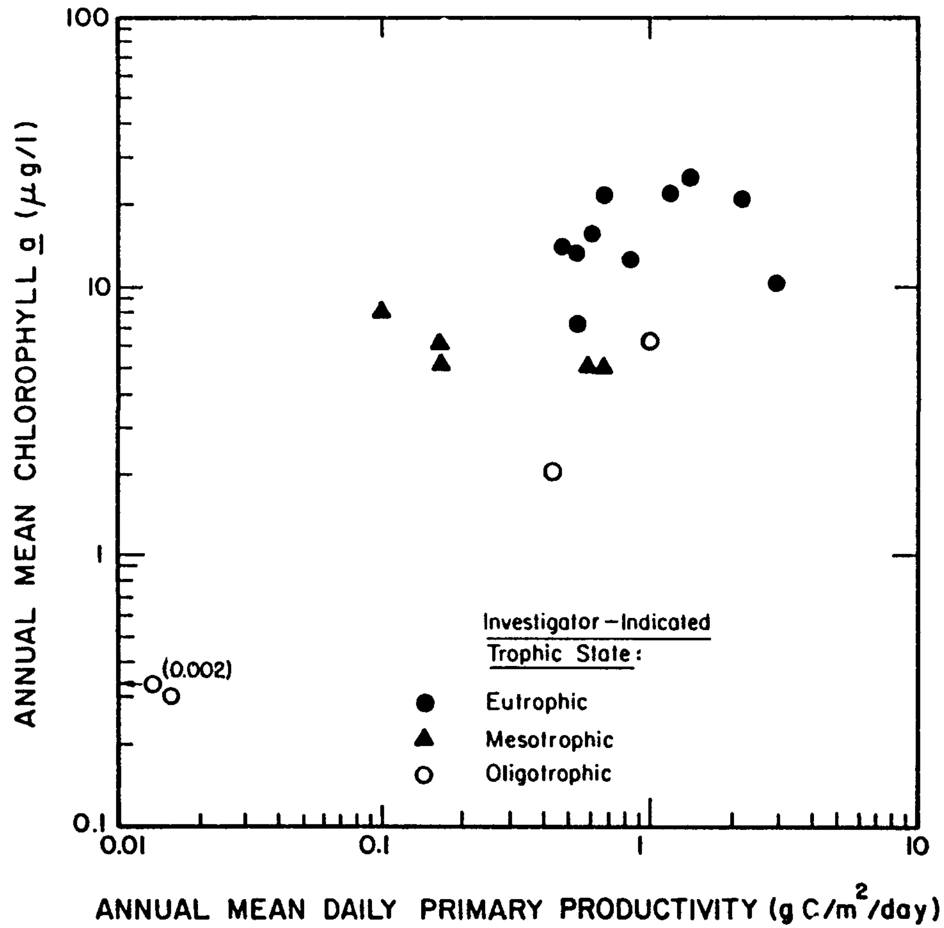

Figure 75. Mean Daily Primary Productivity and Mean Chlorophyll a Relationship in US OECD Water Bodies

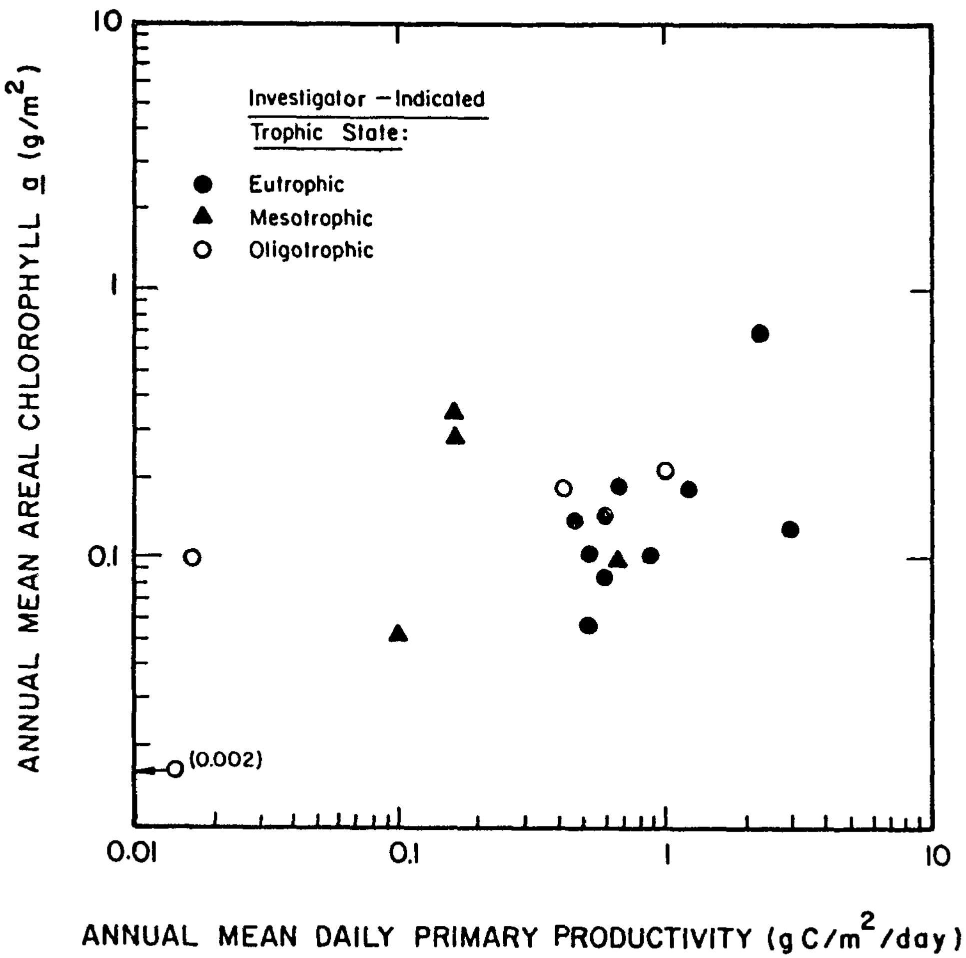

Figure 76. Mean Daily Primary Productivity and Mean Areal Chlorophyll a Relationship in US OECD Water Bodies

constant N:P loading ratio and the basic phosphorus limitation of the water bodies. Several of these correlations, notably total phosphorus versus chlorophyll a and chlorophyll a versus Secchi depth, have been used in the development of several phosphorus load-water quality models presented in the following section of this report.

SECTION XI

APPLICATION OF US OECD RESULTS FOR PREDICTING CHANGES IN WATER QUALITY AS A RESULT OF ALTERING NUTRIENT INPUTS

It is of interest to attempt to predict the change in water quality that might be expected to occur as a result of altering the nutrient loading to a water body. Attention will be focused here on phosphorus loadings for reasons mentioned earlier; namely because many US water bodies are phosphorus-limited, and because phosphorus removal from point sources is both technically and economically feasible (Vollenweider, 1968, 1975a; Lee, 1971, 1973; Vallentyne, 1974; Vollenweider and Dillon, 1974).

The specific question to be addressed is what is the change in water quality expected from a change in the phosphorus loadings to a water body? There are several ways to attempt to answer this question. The best overall approach that can be taken to assess the effects of a change in phosphorus loadings on the trophic conditions of a water body is based on the work of Vollenweider (1975a), discussed in an earlier section of this report. Vollenweider's approach for assessing the degree of fertility of a water body, based on its phosphorus loadings and its mean depth and hydraulic residence time characteristics, was presented graphically in Figure 19. The results of the US OECD eutrophication study, as well as those of the Canadian portion of the North American Project, and the Alpine, Nordic and Shallow Lakes and Reservoirs Project have provided considerable support for this approach. An earlier version of this approach has also been used by the US EPA (1975a) in evaluating the phosphorus loading-eutrophication response in the water bodies in the National Eutrophication Survey, as reflected in their degree of fertility. Further, this earlier version was recommended by the US EPA in their Quality Criteria for Water (US EPA, 1976a) as a basis for determining critical phosphorus loadings for US lakes and impoundments.

As indicated earlier by examination of Figure 19, there is remarkably good agreement between the overall trophic states of the lakes and impoundments in the US OECD eutrophication study as determined by their respective investigators and as indicated by their phosphorus loadings and mean depth/hydraulic residence time characteristics. The US EPA, in the National Eutrophication Survey (US EPA, 1975a), has found a similar agreement for the water bodies that they have investigated thus far. In general, using this relationship, it can be said that in terms of water quality for a given set of morphologic and hydrologic characteristics, as the phosphorus load is increased there is a gradation of deteriorated water quality, as measured by the frequency and severity of obnoxious algal blooms.

The reader should be reminded that the permissible and excessive lines on the Vollenweider phosphorus loading diagram (Figure 19) should not be interpreted as rigid values which define a certain level of water quality. That is, a water body whose phosphorus loading and mean depth/hydraulic residence time characteristics place it just above the excessive line should not be rigidly viewed as having poor water quality. Nor should a water body plotting just below the permissible line be defined strictly as possessing good water quality. Rather, the influence of eutrophication on water quality in a water body is dependent on the public's response, as manifested in an impairment of use of the water body.

As discussed earlier, those water bodies with a given mean depth/hydraulic residence time relationship which plot the greatest vertical distance below the permissible boundary line can be expected to have the best water quality. Conversely, those which plot the greatest vertical distance above the excessive loading line would have the poorest water quality. There is a continual gradient of water quality between these two extremes, with the permissible boundary area defining a general water quality condition acceptable to most of the population.

The position of these lines, as indicated in Equation 11, is influenced by the work of Sawyer (1947). While studying the effects of urban and agricultural runoff on the fertility of 17 lakes in southern Wisconsin, he found a 0.01 mg/l phosphorus concentration in a water body at spring overturn to be a critical concentration for high water quality. Water bodies whose spring overturn phosphorus concentrations exceeded this 0.01 mg P/l critical concentration were likely to experience algal bloom problems during the following summer growing season. The Vollenweider model (Figure 19) is an extension of Sawyer's findings which takes into account some of the morphological and hydrological characteristics of a water body which influence its phosphorus loading-algal growth relationships.

The excessive and permissible phosphorus loading boundary lines on the Vollenweider diagram are based mainly upon the recreational impact of eutrophication. They do not address some of the other parameters of water quality that are influenced by eutrophication. To cite one such example, one could not utilize these phosphorus boundary loading lines to judge whether anoxic conditions would develop in the hypolimnion of a water body. A lake could receive an excessive loading and still have an oxic hypolimnion throughout the year. Dillon (1975; Vollenweider and Dillon, 1974; Dillon and Rigler, 1974b) has reported such an occurrence in a number of water bodies in southern Ontario. This occurred because hypolimnetic oxygen depletion, which is an important eutrophication parameter, is dependent not only on the nutrient load, but also on the hypolimnetic morphology and the hydraulic flushing rate of the water body. Furthermore, the excessive and permissible phosphorus loading lines in Figure 19 do not address the potential eutrophication problems arising from excessive fertilization of water bodies used for domestic water supplies (i.e., taste and odor problems, shortened filter runs, etc.) as contrasted with recreational uses (Gaufin, 1964; DeCosta and Laverty, 1964; Poston and Gamet, 1964; AWWA, 1966).

While the positions of the US OECD water bodies in the Vollenweider diagram (Figure 19) appear to be a good indication of the overall eutrophication and associated water quality for these water bodies, it is desirable to be able to translate this relationship to a eutrophication parameter which is more easily and widely appreciated by both the scientist and layman. For example, in Figure 19, the phosphorus loading to Lake Washington decreased from 2.3 $g/m^2/yr$ in 1964 (water body number 50 in Figure 19), to 0.4 $g/m^2/yr$ in 1971 (number 51 in Figure 19), moving it from the eutrophic zone to a position indicating a much less productive water body. However, a decrease in phosphorus loading or in-lake phosphorus concentration in a water body does not necessarily mean that an improvement in water quality has also occurred. A concomitant change in a parameter which is commonly used to indicate trophic conditions in a water body would help one to appreciate the change in general water quality resulting from a reduced phosphorus input. This section of this report presents the development and application of an approach for assessing changes in water quality to be expected from a change in the phosphorus load to a water body.

The first step in transforming phosphorus loading changes to readily-appreciated indicators of changes in trophic conditions is to examine the relationship between the spring overturn critical phosphorus concentration, and the average chlorophyll concentration during the following summer growing season. Several investigators (Sakamoto, 1966; Dillon and Rigler, 1974a, 1975; Vollenweider, 1976a) have shown a strong relationship exists between these two parameters. Chlorophyll a concentration in a water body is a much more readily observable consequence of phosphorus loading than is a water body's phosphorus concentration. The effects of phosphorus loading can be visibly appreciated as a function of the resulting chlorophyll a concentration or "greenness" of a water body. Vollenweider (1976a) has plotted chlorophyll a concentrations as a function of the phosphorus loading characteristics of a water body. The theoretical basis of this approach was presented in an earlier section of this report. The reader is reminded that the phosphorus loading characteristic expression, $((L(P)/q_s)/(1+\sqrt{\bar{z}/q_s}))$, is equivalent to the predicted in-lake steady state total phosphorus concentration (Equation 20). Since this relationship is based partly on Sawyer's theoretical critical phosphorus concentration at spring overturn, one could also use this phosphorus loading characteristics and chlorophyll _a_ concentration relationship to predict the chlorophyll _a_ concentration expected in a water body for a given phosphorus loading, as modified by its hydraulic loading. Vollenweider (1976a) has used this relationship in this manner with good results.

The US OECD phosphorus loading and chlorophyll _a_ data have been used in a similar manner to determine if the US OECD water bodies follow the same pattern and to evaluate the ability of Vollenweider's phosphorus loading characteristics and chlorophyll _a_ relationship in assessing the relative trophic condition of water bodies. This was illustrated in Figure 22. The US OECD data base, like Vollenweider's, consisted of a mixed collection of summer and annual mean chlorophyll values. However, the correlation is unquestionable. Using the method of least squares, the regression line through this double logarithm plot (Figure 22) was determined to be:

$$\log_{10} [\text{chlorophyll } \underline{a}] = 0.760 \log_{10} [(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})] - 0.259 \tag{37}$$

or this regression line, r = 0.77. This is compared to Vollenweider's (1976a) correlation coefficient of 0.87. If only the annual mean chlorophyll _a_ values are used, the regression equation becomes:

$$\log_{10} [\text{chlorophyll } \underline{a}] = 0.709 \log_{10} [(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})] - 0.173 \tag{38}$$

For this regression equation, r = 0.78.

It should be noted that there were 43 data sets of annual mean chlorophyll _a_ and $(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$ values versus only 9 data sets of growing season values. Thus, the regression equations are weighted heavily for the data sets of annual mean values. The differences between the growing season mean and annual mean chlorophyll _a_ values as they affect the relationship illustrated in Vollenweider's chlorophyll _a_ and phosphorus loading characteristics plot are not being addressed in this present analysis. However, as this relationship is a double logarithmic plot, this would indicate that a very large change in phosphorus load would be necessary to bring about a substantial change in the chlorophyll _a_ concentration. Vollenweider (1976a) has presented support for use of this relationship by plotting the phosphorus loading characteristics and chlorophyll _a_ concentrations measured in Lake

Washington, both before and after the completion of its extensive sewage diversion program. The resultant chlorophyll a changes tracked quite closely the expected changes (See Figure 9 in Vollenweider, 1976a). It is noted, however, that a few years after the sewage diversion project was completed, the chlorophyll a concentrations tended to be higher than that based on Vollenweider's relationship between phosphorus loading and chlorophyll. As discussed in an earlier section, this is most probably related to the fact that a water body takes several years to adjust to a new phosphorus loading. This time period of adjustment is equal to approximately three phosphorus residence times (Sonzogni et al., 1976). The final result in Vollenweider's (1976a) application of the Lake Washington data to his phosphorus loading and chlorophyll a relationship is in accordance with what is expected based on the phosphorus load under equilibrium conditions.

Thus, in summary, examination of the US OECD data as presented in Figure 22 indicates there is good agreement between overall chlorophyll levels and phosphorus loads (as expressed in the phosphorus loading characteristics term $(L(P)/q_s)/(1+\sqrt{z/q_s})$) for water bodies studied in the US OECD eutrophication study. This relationship will be used in the following pages to further develop a meaningful relationship for assessing changes in water quality to be expected following a change in the phosphorus load to a water body.

The next step in this effort is to examine the relationship between chlorophyll and Secchi depth and then to unite this relationship with the phosphorus load to a water body. As indicated in an earlier section, the use of the Secchi depth of a water body as an indicator of its algal biomass and overall water quality has been proposed by several investigators (Edmondson, 1972; Carlson, 1974; Shapiro, 1975b; Shapiro et al., 1975). Indeed, the Secchi depth is thought to be one of the best overall parameters that the public could respond to for improved water quality. Edmondson (1972) and Shapiro et al. (1975b) have presented similar conclusions on the value of the Secchi depth as a measure of the impairment of water quality by excessive fertilization. A number of investigators have demonstrated an inverse non-linear relationship between the chlorophyll a content of a water body and its Secchi depth (Edmondson, 1972; Carlson, 1974; Bachmann and Jones, 1974; Dillon and Rigler, 1974a; 1974b; Dobson, 1975; Norvell and Frink, 1975 and Michalski et al., 1975). Further, the US OECD data showed a similar relationship. The pertinent data are presented in Table 27. A plot of the chlorophyll a concentrations and Secchi depths from these various sources is presented in Figure 77. The data reported by Edmondson (1972), although extensive and extending over a number of years, was not included in this plot since this relationship was for only one water body while the other data sets were from a variety of water bodies. It was felt by these reviewers that his data

Table 27. DATA FOR CHLOROPHYLL a AND SECCHI DEPTH RELATIONSHIP[a]

Carlson (1974)[a]		Dobson (1975)[a]		Dillon & Kigler (1974a)[a]		Bachmann & Jones (1974)[a]		Norvell & Frink (1975)[a]		US OECD Eutrophication Study[a]		Michalski et al. (1975)[a]	
Chlor a[b]	SD	Chlor a[c]	SD	Chlor a[c]	SD	Chlor a	SD	Chlor a	SD	Chlor a	SD	Chlor a	SD
0.94	8	5.9	2.55	1.6	4.3	4.5	5.5	0.6	6.3	14.6[d]	3.6	1.1	8.05
2.6	4	5.2	3.16	1.4	5.5	5.5	5.1	0.9	8.2	5.9[e]	1.5	2.2	6.25
6.4	2	4.8	2.5	1.2	5.9	1.5	4.75	50	1.8	6.0[e]	2.1	1.0	4.35
20	1	4.8	2.4	1.3	6.0	3.0	3.7	30	1.5	6.3[d]	2.0	2.3	4.0
56	0.5	4.6	2.8	1.8	5.5	5.0	3.5	14	2.2	5.0	1.7	7.2	3.35
1	7.7	4.2	3.7	1.1	6.4	4.0	3.45	7.2	4.0	6	2.3	8.0	3.15
2	4.8	4.0	3.75	0.8	8.7	3.5	3.0	13.9	2.5	5	2.3	6.8	3.0
3	3.6	3.8	5.0	1.0	4.9	4.5	2.9	34	2.0	7.4	1.8	13.5	2.65
4	3.0	3.8	4.6	1.1	5.2	8.0	2.9	2.4	5.7	5.6	2.4	6.6	2.5
5	2.6	3.6	6.0	1.5	5.3	4.5	2.85	7.5	4.5	20[e]	1.8	4.1	2.1
6	2.3	1.3	7.2	1.2	5.5	5.0	2.5	3.6	5.3	26.5[d]	1.5	9.3	2.1
7	2.0	1.1	8.1	1.0	5.4	6.0	2.5	0.6	5.0	6	2.7	14.1	1.95
8	1.9	1.1	8.7	1.8	6.2	6.0	2.4	1.5	6.0	4	2.5	5.8	1.80
9	1.7	0.7	7.7	1.1	6.1	9.0	2.3	4.8	5.0	33.9[d]	0.8	14.4	1.1
10	1.6	1.3	7.9	2.1	5.9	12.0	2.1	9.0	3.5	3.5[d]	2.4	20.0	1.1
12	1.4	0.7	8.6	0.95	6.3	9.5	2.05	4.8	4.8	53[d]	1.0	24.9	0.85
14	1.3			2.7	3.5	10.0	1.9	7.8	3.3	13.2	1.4	18.0	0.45
16	1.2			1.0	10	10.0	1.8	38	1.9	21.2	1.2		
18	1.1			0.4	8.55	12.5	1.7	5.2	3.5	6	1.8		
20	1.0			0.9	8.1	10.5	1.65	2.8	4.8	3	2.2		
0.5	12.3			0.4	7.45	22.5	1.6	2.0	4.5	5	3.1		
22.5	0.90			0.7	6.85	30.5	1.45	4.2	3.8	2	3.0		
25.0	0.85			0.7	6.4	14.0	1.2	5.6	4.0	10	3		
27.5	0.80			2.0	6.1	23.0	1.1	14.3	2.5	5	2.3		
30.0	0.75			1.3	6.2	14.5	1.0	13.8	3.0	21	1.5		
35.0	0.70			1.7	5.95	24.0	1.0	1.7	4.5	12	1.8		
40.0	0.65			1.4	5.7			3.2	4.3	10	1.8		

Table 27 (continued). DATA FOR CHLOROPHYLL a AND SECCHI DEPTH RELATIONSHIP[a]

Carlson (1974)[a]		Dobson (1975)[a]		Dillon & Rigler (1974a)[a]		Bachmann & Jones (1974)[a]		Norvell & Frink (1975)[a]		US OECD Eutrophication Study[a]		Michalski et al. (1975)[a]	
Chlor a[b]	SD	Chlor a[c]	SD	Chlor a[c]	SD	Chlor a	SD	Chlor a	SD	Chlor a	SD	Chlor a	SD
45.0	0.60			2.4	5.7			15.5	2.0	90	0.6		
50.0	0.55			0.5	5.5			3.2	6.8	65	0.8		
60.0	0.50			1.5	5.5			2.3	7.5	15	1.5		
70.0	0.45			1.5	5.35			3.2	6.0	12.8[d]	1.6		
				1.5	5.25			31	2.5	5	3.3		
				1.7	5.25			9.9	3.0	15	2.3		
				2.2	5.25			5.9	4.5	31	1.7		
				2.2	4.95			5.3	3.5	12.3[d]	1.4		
				6.6	4.7			5.5	3.3	0.3	28.3		
				1.2	4.7			2.4	6.0	21	2.1		
				1.8	4.7			13.1	2.3	26	1.6		
				2.4	4.7			9.0	3.2	22	2.3		
				1.0	4.55			1.8	6.8	28	1.9		
				3.4	4.55			1.2	7.2	27	1.7		
				1.3	4.45			0.7	7.3	40	2.2		
				2.2	4.35			2.4	8.2	23	2.8		
				3.8	4.35			103	1.0	28	2.3		
				0.8	4.25			25	2.5	19[d]	1.5		
				1.4	4.25					29[d]	1.2		
				1.8	4.1					0.3	28.0		
				2.5	3.95					12	2.2		
				3.3	3.85					20	1.2		
				2.7	3.75					6	3.5		
				1.7	3.7					8	1.9		
				2.8	3.6					4	1.9		
				1.4	3.55								
				1.5	3.45								

Table 27 (continued). DATA FOR CHLOROPHYLL *a* AND SECCHI DEPTH RELATIONSHIP[a]

Carlson (1974)[a]		Dobson (1975)[a]		Dillon & Rigler (1974a)[a]		Bachmann & Jones (1974)[a]		Norvell & Frink (1975)[a]		US OECD Eutrophication Study[a]		Michalski *et al.* (1975)[a]	
Chlor *a*[b]	SD	Chlor *a*[c]	SD	Chlor *a*[c]	SD	Chlor *a*	SD	Chlor *a*	SD	Chlor *a*	SD	Chlor *a*	SD
				2.9	3.45								
				2.2	3.4								
				3.2	3.35								
				5.4	2.65								
				7.9	2.35								
				1.7	2.25								
				4.9	2.2								
				9.0	2.15								
				6.1	2.15								
				5.4	2.0								
				13.2	1.9								
				17.9	1.7								
				3.9	1.4								
				19.6	1.2								
				14.5	1.2								
				14.9	1.05								
				7.5	1.0								
				13.1	0.75								
				16.3	0.45								

Explanation:

[a]Source of data; all chlorophyll values are in μg/l; all Secchi depth values are in meters

[b]Surface chlorophyll

[c]Summer total chlorophyll

[d]Upper 2 meters of water column

[e]Summer surface mean values

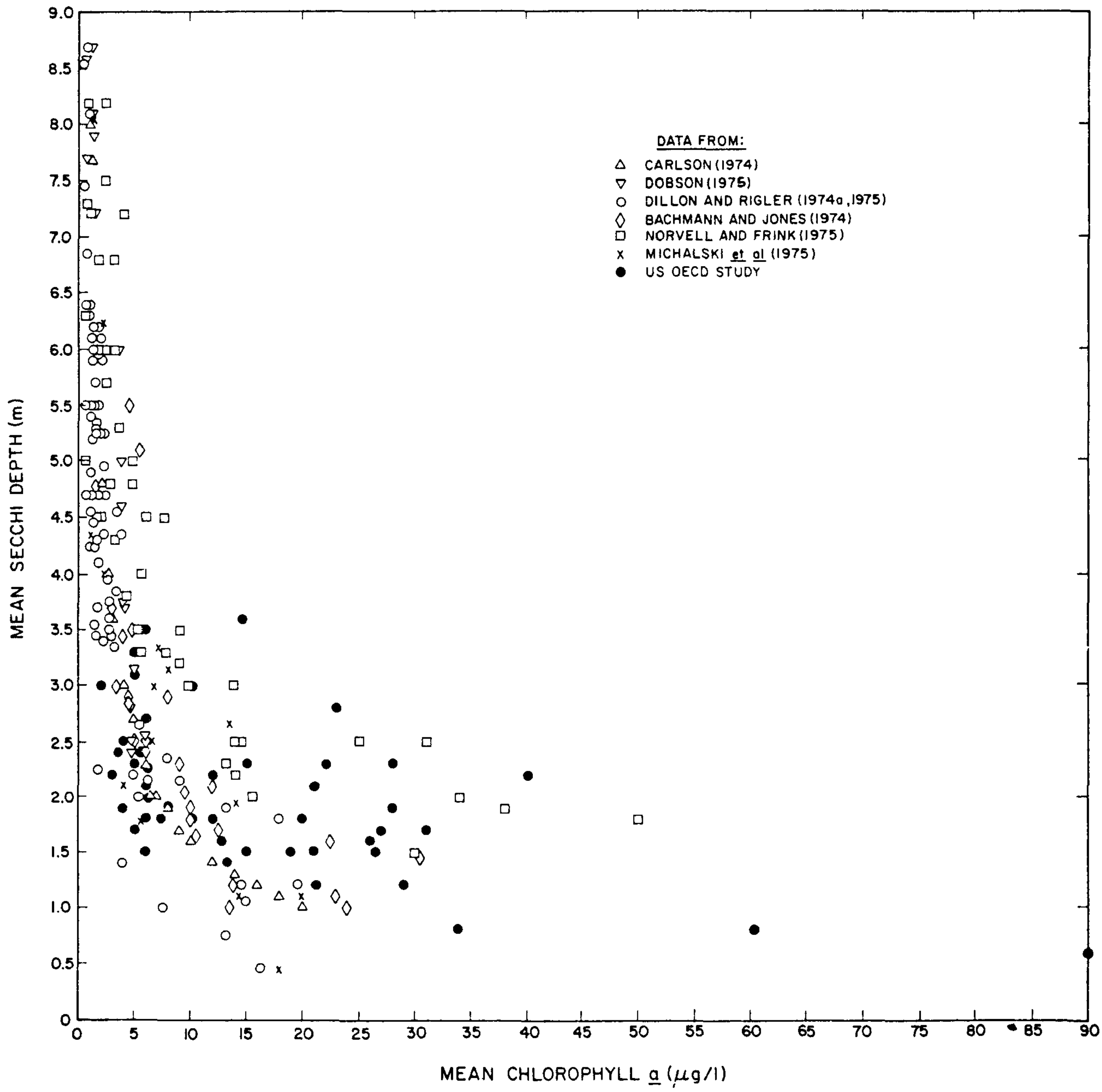

Figure 77. Secchi Depth and Chlorophyll a Relationship in Natural Waters (Linear Scale).

base would bias the resultant plot toward the chlorophyll a and Secchi depth relationship typical of Lake Washington. In addition, an examination of the Secchi depth and chlorophyll relationship in Lake Washington showed it to be somewhat different than that seen with the other sources listed above. Consequently, only data reported from a wide range of water body types were used in Figure 77. However, a comparison of this figure with the plot presented by Edmondson (1972) shows a similar hyperbolic relationship between these two parameters, with the slope of the curve steepest at the lower chlorophyll concentrations.

As it is difficult to get an accurate regression line of best fit for the non-linear chlorophyll a and Secchi depth relationship illustration in Figure 77, the same data sets were plotted on a double logarithm plot. This is illustrated in Figure 78. The regression equation for this plot is:

$$\log_{10} \text{Secchi depth} = -0.473 \log_{10} [\text{chlorophyll } \underline{a}] + 0.803 \tag{39}$$

The regression line has a correlation coefficient, $r = -0.85$, indicating a good correlation between the chlorophyll a content and Secchi depth in natural waters for a wide variety of water bodies located throughout the US.

Since the chlorophyll content of a water body is related to its phosphorus loading characteristics (Figure 22), and since a strong correlation was demonstrated above between chlorophyll a and Secchi depth (Figure 78), there should also be a relationship between a water body's phosphorus loading characteristics and its Secchi depth. In fact, the final step remaining in this exercise is to unite both these relationships (Figures 22 and 78) into a single expression which directly relates these two parameters. This has been accomplished by producing a double logarithmic plot of the phosphorus loading expression, $(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$, and Secchi depth, as illustrated in Figure 79. The line of best fit was extrapolated from the data presented in Figures 22 and 78. Chlorophyll a values, as a function of a water body's phosphorus loading characteristics, were taken from Figure 22. Then, the expected Secchi depth for a given chlorophyll a concentration was taken from Figure 78. The expected Secchi depth was then plotted as a function of the original phosphorus loading expression above, to produce the line of best fit illustrated in Figure 79. Using least square analysis, the regression equation for this line is:

$$\log_{10} \text{Secchi depth} = -0.359 \log_{10} [(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})] + 0.925 \tag{40}$$

Using this relationship (Figure 79), one can determine the Secchi depth to be expected as a function of the phosphorus loading

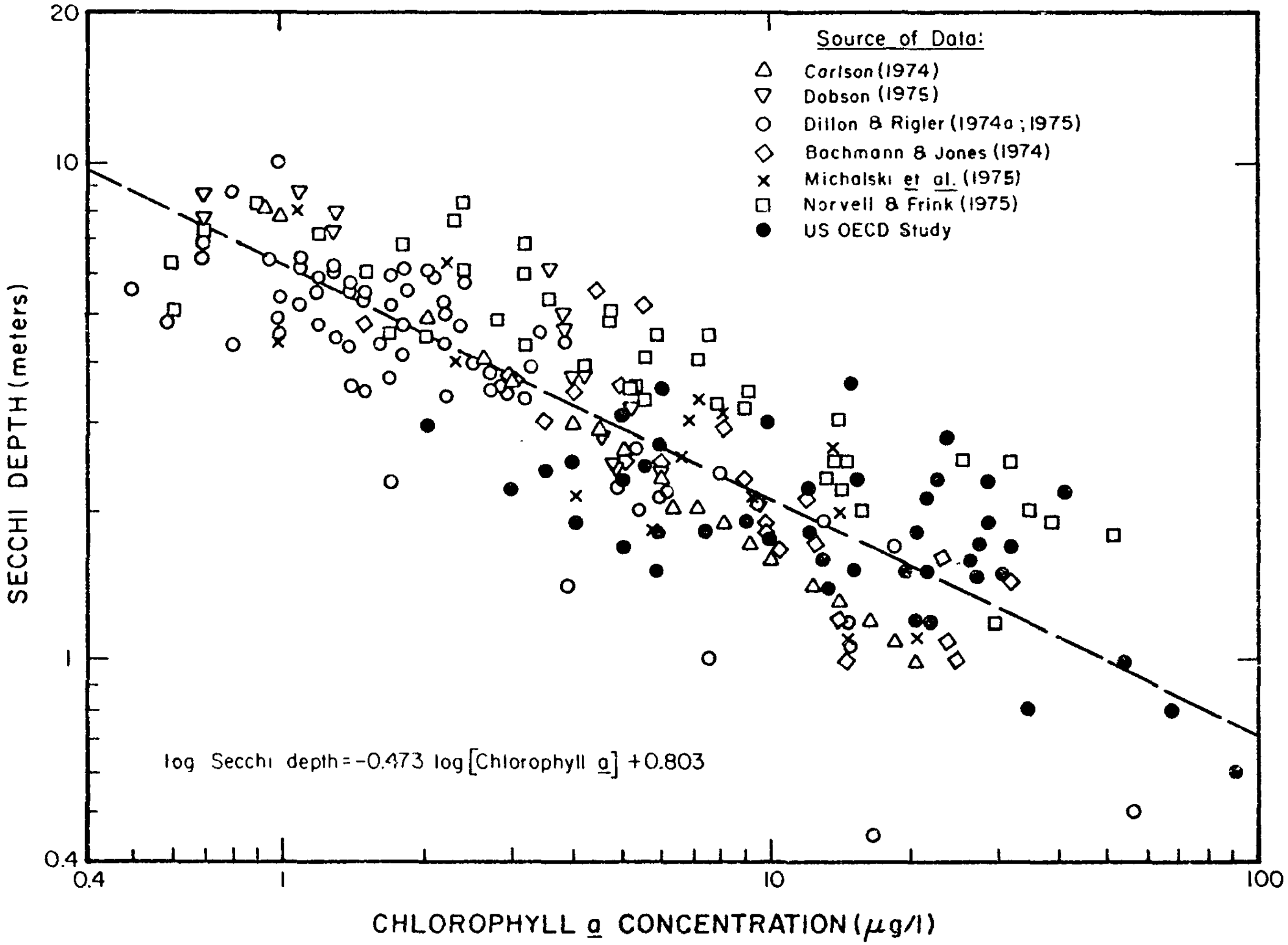

Figure 78. Secchi Depth and Chlorophyll a Relationship in Natural Waters (Log-Log Scale).

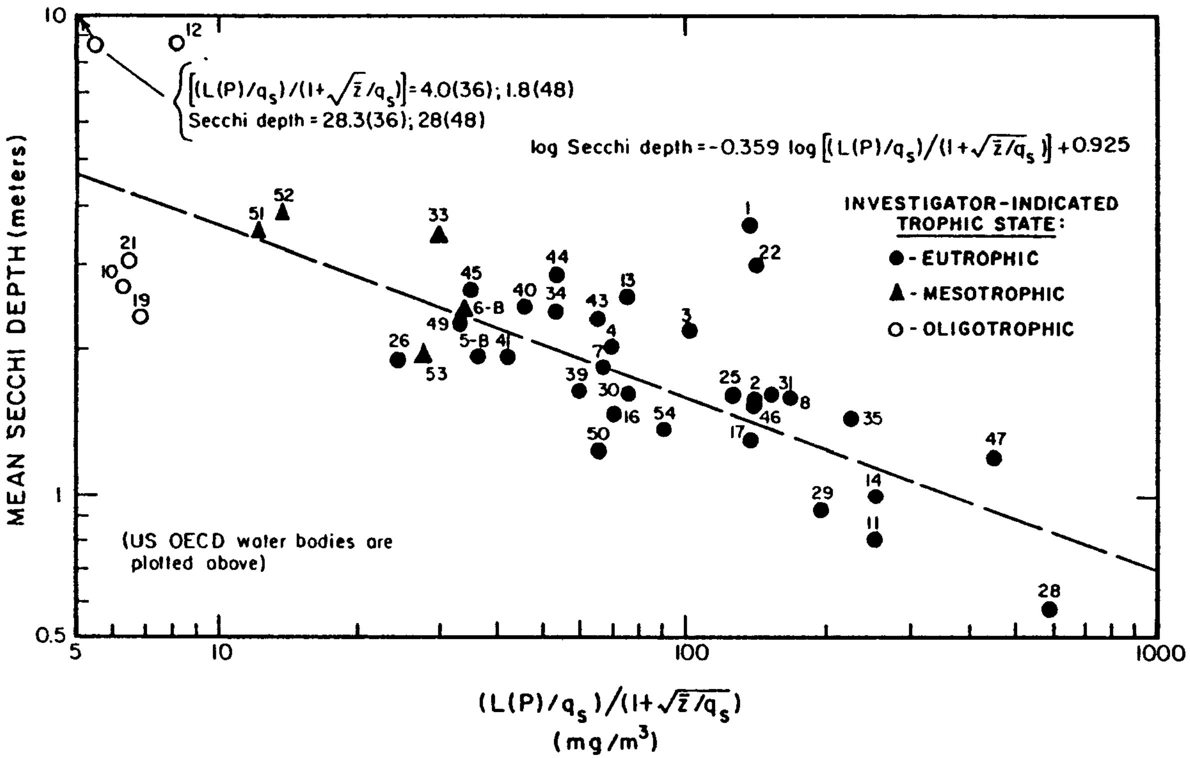

Figure 79. Phosphorus Loading Characteristics and Secchi Depth Relationship in Natural Waters

characteristics of a given water body. As indicated earlier, this relationship allows one to be able to determine the change in water quality in a water body, expressed as a function of its Secchi depth, which would result from a change in its phosphorus load. The change may be deterioration or enhancement of water quality (i.e., decrease or increase in Secchi depth) depending on whether the phosphorus flux to a water body was increased or decreased. This relationship, therefore, represents a single, practical application of some of the results of the US OECD Project in assessing the effects of phosphorus loadings to water bodies as expressed as a function of a widely-appreciated parameter of eutrophication, both to scientists and laymen.

There are several precautions that should be noted in the use of this relationship. One consideration is that it would hold only for those water bodies where the primary factor controlling water clarity is phytoplankton. It would not be applicable in its present form to water bodies with large amounts of inorganic turbidity or color. However, it may be possible to partially correct for the effects of excessive inorganic turbidity and color on the Secchi depth of a water body. According to Vollenweider (1977), in the simple case, the Secchi depth may be computed as the integral of the turbidity above the Secchi disk (i.e., $\int_0^S T(z)dz$ = constant, where S = Secchi depth (m), $T(z)$ = mineral turbidity at depth z (mg/l), and $T(z)$ is inhomogenous over depth z). For the homogenous case, the Secchi depth may be calculated as $S = 1/(k_1 + k_2 (C) + k_3 (T) + k_4 (Chl))$, where C = color (mg P_t/l), T = mineral turbidity (mg/l), Chl = chlorophyll a (mg/m^3) and k_1, k_2, k_3 and k_4 = constants. Vollenweider (1977) is evaluating the constants as follows: $k_1 \approx 0.025$, $k_2 \approx 0.005$ to 0.01 and $k_4 \approx 0.01$ to 0.02. The constant k_3 is difficult to estimate because of the lack of appropriate data for expression of the interaction of primarily biological turbidity with chlorophyll a. In relatively transparent water (i.e., little color or mineral turbidity), one may approximate k_3 by use of the relationship, $S = 1/(k_3(Chl)$. For very transparent waters, C, T and Chl can be expected to be very small. Accordingly, recalling $k_1 \approx 0.025$, the Secchi depth approximates 40 meters (i.e., S = 1/0.025). For less transparent waters, one will have a family of curves, depending mainly on the terms k_2 (C) and k_3(T). Thus, one could attempt to correct the Secchi depth for high color or inorganic turbidity in water bodies using the relationships expressed above. The corrected Secchi depth can likely then be applied in the previously-mentioned equations relating the phosphorus loading, chlorophyll a and Secchi depths in natural waters. It should be noted, however, that it was not possible to test the homogenous equation above because of lack of sufficient data for the US OECD water bodies. Few water bodies in the US OECD eutrophication study had excessive color or turbidity to permit such an evaluation.

This relationship would also not hold for water bodies whose excessive phosphorus loadings were manifested principally in excessive macrophyte growths and attached algae, rather than in nuisance planktonic algal blooms. Such water bodies tend to have a larger Secchi depth than would be expected on the basis of phosphorus loadings alone since a portion of the phosphorus input would be incorporated into the macrophytes rather than into the phytoplankton. Finally, this relationship would hold only for those water bodies whose phosphorus loadings were relatively constant (i.e., in an equilibrium state). This is because the phytoplankton populations and, hence, chlorophyll content of a water body, are a function of the phosphorus concentration, which in turn is a function of the phosphorus loading to the water body. The relationship between the phosphorus concentration in a water body and the phosphorus load to the water body is a complicated one (Vollenweider, 1968), being a function of the water body's mean depth, hydraulic residence time, internal loading, aquatic plant population, etc. However, if the phosphorus load is relatively constant over the annual cycle, it can be expected that the mean total phosphorus concentration is also relatively constant over the annual cycle. Under such equilibrium conditions, the use of Figure 79 to predict Secchi depth as a function of a water body's phosphorus loading characteristics should present no problems. On the other hand, if the phosphorus load to a water body is increased or decreased significantly, as in a sewage diversion project or the introduction of sewage treatment plant effluent to a water body, then the relationship expressed in Figure 79 would likely not be valid for prediction of Secchi depth. As discussed by Sonzogni et al. (1976), a water body does not instantaneously adjust to a new phosphorus load. Rather, a period of approximately three times the phosphorus residence time is necessary for a water body to adjust to a new phosphorus load. After this time period, assuming the phosphorus load has not been further changed since an initial increase or decrease, one could expect to again be able to use Figure 79 to predict Secchi depth in a water body as a function of its phosphorus load characteristics. It should be noted that this represents a simple, quantitative and practical methodology for determining what the expected Secchi depth will be in a water body in response to a sewage diversion or advanced treatment project, prior to initiation of the project.

If one examines the phosphorus loading and Secchi depth data for Lake Washington (Vollenweider, 1976a; Edmondson, 1975a), the 1964 phosphorus loading for Lake Washington, at the initiation of its sewage diversion project, gives it an $(L(P)/q_s)/(1+\sqrt{z/q_s})$ value of approximately 100 (Vollenweider, 1976a). This corresponds to a Secchi depth of about 1.6 m. Edmondson reported a mean Secchi depth for Lake Washington in 1964 to be 1.2 m. However, the phosphorus loading had been increasing dramatically since 1957 (i.e., $(L(P)/q_s)/(1+\sqrt{z/q_s})$ value of approximately 40 in 1957 and 1964 value of 100), and consequently the relatively poor prediction of Secchi depth was not unexpected. However, as noted earlier in Table 21, the phosphorus residence time for Lake Washington was approximately one year in 1964. Thus, according to Sonzogni et al. (1976), one could expect a new phosphorus concentration equilibrium condition in about three years. However, the sewage diversion project, although begun in the early 1960's, was not completed until about 1968. Therefore, by 1972-1973 at the latest, one could expect phosphorus equilibrium conditions to again exist.

In fact, if one examines the phosphorus loading expression value for Lake Washington in 1971 and 1974 (Vollenweider, 1976a) and compares the Secchi depth predicted in Figure 79 (i.e., 3.7 m and 3.5 m, respectively) with the mean Secchi depth reported by Edmondson (i.e., 3.5 m and 3.8 m, respectively), they are quite similar. The small discrepancies may exist because Vollenweider (1976a) appeared to use slightly different phosphorus loadings in his Lake Washington calculations than those reported by Edmondson (1975a), at least for 1971. If one uses the phosphorus loadings reported by Edmondson in 1971 in Figure 79, the predicted and reported Secchi depths for that year are identical. Edmondson (1975a) did not report phosphorus loadings for 1974, so it was not possible to compare the predicted and observed Secchi depths for that year based on his loadings. For this reason, the $(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$ expression values indicated by Vollenweider (1976a) were used to compare the predicted and observed Secchi depth values for 1971 and 1974. Even so, the agreement between these two Secchi depth values for 1971 and 1974 is quite good, lending support to this approach in assessing water quality as a function of several easily understood and measurable parameters. It should be noted that, as was the case for the phosphorus loading characteristic and chlorophyll a concentration relationship, this new relationship also indicates that a relatively large change in the phosphorus load must occur to water bodies in order to show marked improvement in water clarity.

It is also feasible to develop a model which relates phosphorus loads to the water quality parameter of hypolimnetic oxygen depletion. This latter parameter is of concern because of its implications for the development of anoxic conditions in hypolimnetic waters, especially in eutrophic water bodies. The consequences of anoxic conditions in the hypolimnion on the cold water fisheries which usually populate this region of a water body are obvious. The chemically-reducing conditions usually found in an anoxic hypolimnion also have implications for water quality. For these reasons, the development of a water quality model relating phosphorus loads to hypolimnetic oxygen depletion is discussed below.

Gilbertson et al. (1972) found a remarkably good linear correlation between municipal phosphorus loads and hypolimnetic oxygen depletion rates in the central basin of Lake Erie. Based on the observed period of thermal stratification and the oxygen levels in Lake Erie's central basin, Gilbertson et al. determined that the critical oxygen depletion rate in the hypolimnion of Lake Erie's central basin was about 2.7 mg O_2/l/month. That is, a hypolimnetic oxygen depletion rate of 2.7 mg O_2/l/month during the period of thermal stratification would produce a zero concentration of oxygen in the hypolimnion of the central basin of Lake Erie by the end of a given summer. Examination of the historical data for Lake Erie (Gilbertson _et al_., 1972) indicates this critical depletion rate corresponds to the 1955 phosphorus loading conditions of about 12,000 tons per year, and has been exceeded every year since that time.

The observations of Gilbertson _et al_. suggest that a generalized approach relating phosphorus loads and hypolimnetic oxygen depletion would appear to be feasible for a wide range of water bodies. One approach for developing such a relationship involves the use of a model derived by Lasenby (1975) between areal hypolimnetic oxygen depletion and Secchi depth. Studying 14 lakes in southern Ontario, and several other water bodies Lasenby reported that a strong inverse relationship (r=-0.85) appeared to exist between the areal hypolimnetic oxygen depletion rate and Secchi depth in these water bodies, as follows:

$$\log_{10} \text{ areal hypolimnetic oxygen depletion rate (mg } O_2/cm^2/\text{day)}$$

$$= -1.37 \log_{10} \text{ Secchi depth (m)} -0.65 \qquad (41)$$

An assumption in Lasenby's model was that the quantity of seston sinking into the hypolimnion was proportional to the quantity in the epilimnion. Lasenby (1975) has indicated that the linear development of his hypolimnetic oxygen depletion model suggests that hypolimnetic oxygen consumption is not too sensitive to brief changes in productivity and, therefore, relatively few measurements should give a good estimate of oxygen depletion rates.

With the relationship expressed earlier between phosphorus loading and growing season mean Secchi depth (Equation 40), and using Secchi depth as the common variable, Equation 41 above was used to derive a relationship between phosphorus loading and areal hypolimentic oxygen depletion. This model was then tested using US OECD data, as well as data presented by Welch and Perkins (1977) for a large number of water bodies with a wide range of trophic conditions. Examination of the data indicated that Lasenby's relationship, derived mainly from oligotrophic and mesotrophic water bodies, tended to overestimate the areal hypolimnetic oxygen depletion rates in the majority of the water bodies. Consequently, it was decided to use simple linear regression techniques, as was done with Figure 22, to determine the best relationship. The following regression was obtained:

$$\log_{10} \text{ areal hypolimnetic oxygen depletion rate (g } O_2/m^2/\text{day)}$$

$$= 0.467 \log_{10} [(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})] -1.07 \qquad (42)$$

This relationship is illustrated in Figure 80, along with the available US OECD data, as well as data furnished by Welch and

Perkins (1977) for several other water bodies. The model refers to the mean areal hypolimnetic oxygen depletion rate during the period of thermal stratification. Since the oxygen depletion rate is expressed on an areal basis, it can be applied to any hypolimnetic volume, regardless of size or oxygen content. It should be noted that the units of hypolimnetic oxygen depletion in Equation 42 are different from those presented in Equation 41.

Very few studies on hypolimnetic oxygen depletion were conducted on the US OECD water bodies. Consequently, the data base for testing this phosphorus load-hypolimnetic oxygen depletion model (Equation 42) was not as large as that for either the phosphorus load and chlorophyll a or Secchi depth models. Although there is some scatter of the data in Figure 80, considering the uncertainty in the data available for the phosphorus loads, hypolimnetic volume, area and oxygen concentration, thermal stratification, etc., the agreement between the predicted and observed values is reasonably good and provides support for this model as a predictive management tool for assessing the effects of a given phosphorus load on the hypolimnetic oxygen depletion in a water body. Further details concerning this model are presented in Rast (1977).

APPLICATION OF RESULTS FOR ASSESSING WATER QUALITY IN LAKES AND IMPOUNDMENTS

The approach presented in this section of this report can be used to assess the potential effects of phosphorus load reductions on water quality in lakes and impoundments. Assessments of this type are becoming increasingly important in developing the most cost-effective phosphorus control strategies for these water bodies. In the past, eutrophication control strategies were frequently based on the removal of phosphorus from its most readily controllable sources, without any quantitative assessment possible beforehand of the magnitude of water quality improvement that would result from controlling the phosphorus input to a certain degree. The implementation of Section 314-A of PL 92-500 will require water pollution regulatory agencies throughout the US to develop nutrient control strategies for those water bodies which are found to be excessively fertile. As a result of the US OECD eutrophication program, it will now be possible for these agencies to quantitatively assess the magnitude of water quality improvement that can be achieved as a result of a phosphorus input reduction of a certain amount. This section of this report discusses the application of these results to a hypothetical situation which is likely to be typical

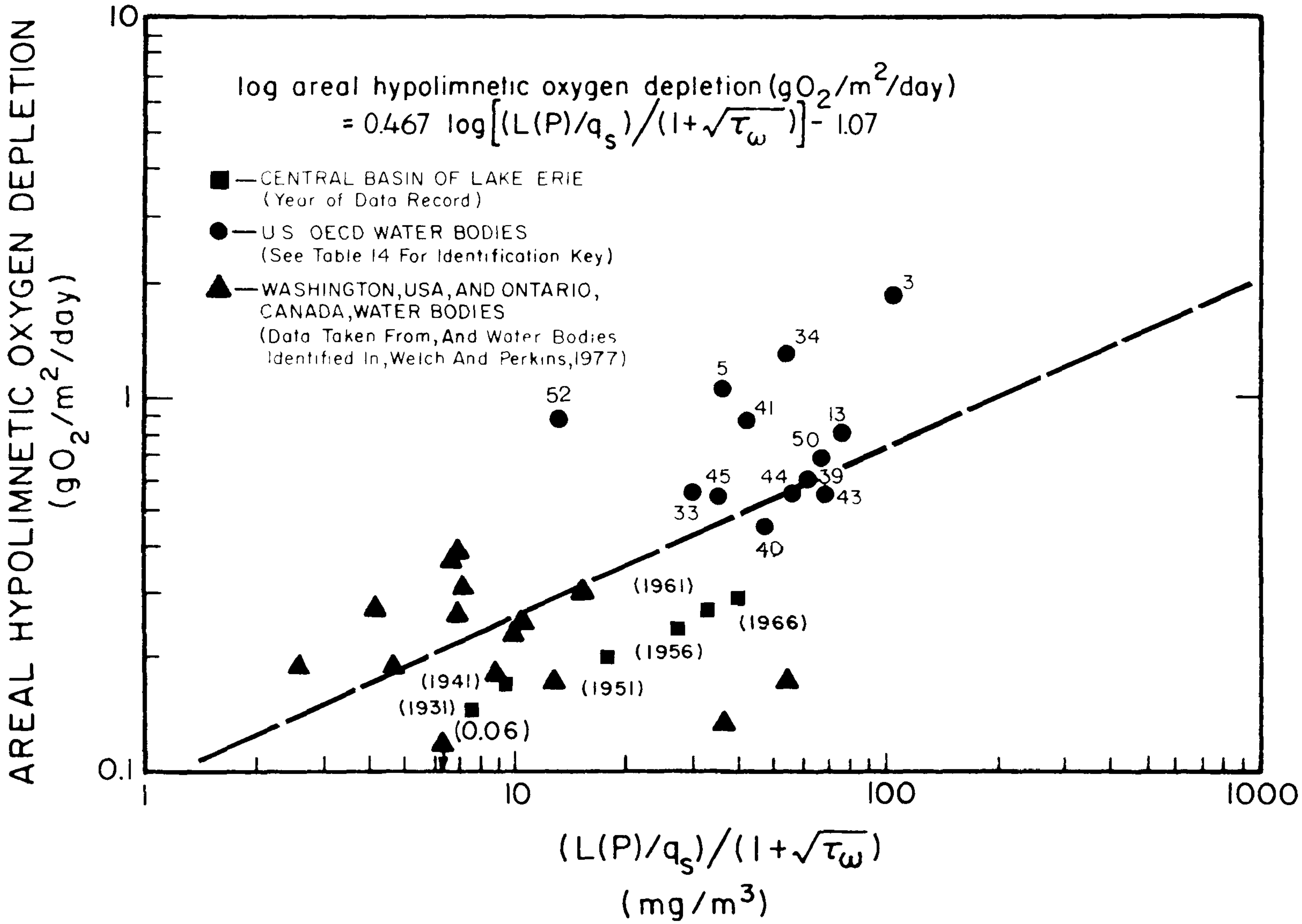

Figure 80. Phosphorus Loading Characteristics and Hypolimnetic Oxygen Depletion Relationship in Natural Waters

of what pollution control agencies will encounter as they attempt to implement Section 314-A of PL 92-500. The approach taken in this section is patterned after the approach developed by Rast (1977) and used by Lee (1976) and Lee et al. (1977) to assess the improvement in water quality that would occur in the Great Lakes as a result of a phosphate detergent ban in the State of Michigan.

A hypothetical phosphorus loading situation has been conceived in this section to illustrate the use of the approach described above for assessing the potential effects of phosphorus load reductions on water quality in a water body. Several phosphorus load reduction possibilities are considered. The phosphorus loads and other pertinent data for analyzing the potential effects of phosphorus load reductions are summarized in Table 28. In this hypothetical water body, the point source inputs are 56 percent of the total phosphorus load, with non-point sources comprising the other 44 percent. The initial phosphorus loading is the hypothetical load for 1975 and consists of both point sources (domestic wastewater treatment plants) and non-point sources (land runoff and atmospheric inputs). As shown in Table 28, in this hypothetical example, the point source phosphorus load is 6.6 million kg/yr, while the nonpoint phosphorus load is 6.2 million kg/yr. For an assumed surface area of 2.6×10^{10} m^2, this corresponds to an areal loading of 0.46 g $P/m^2/yr$. The first modified phosphorus loading considers the effects of a detergent phosphate ban on the loading to the water body. It was determined by Lee (1976) that a detergent phosphate ban would result in approximately a 35 percent reduction in the amount of phosphorus in domestic wastewaters. This percentage value was used in these examples. This would reduce the phosphorus input to the hypothetical water body from this source by the same magnitude. This reduction would change the point source phosphorus load from 6.6×10^6 kg/yr to 4.3×10^6 kg/yr, and reduce the overall areal load from 0.46 to 0.37 g $P/m^2/yr$.

The second modified condition considers the effects of a 90 percent phosphorus removal from the domestic wastewater treatment plant loadings. In this case, it will simulate the effects of advanced waste treatment for phosphorus removal on the point source load to the water body. The 90 percent removal reduces the point source phosphorus load to 6.6×10^5 kg/yr, and the overall areal load to 0.23 g $P/m^2/yr$. The third modified phosphorus loading simulates the effect of advanced waste treatment on the sewage treatment plant inputs plus phosphorus loading reduction from the non-point sources. In this case, it will simulate the effects of advanced waste treatment phosphorus removal techniques on land runoff.

Figure 81 presents the Vollenweider phosphorus load and mean depth/hydraulic residence time relationship for this hypothetical water body. Included in Figure 81 are the expected changes in phosphorus loading for each of the phosphorus loading reduction scenarios described above. Examination of Figure 81 shows that the phosphorus load for 1975 places the water body in the eutrophic

Table 28. SUMMARY OF DATA FOR HYPOTHETICAL WATER BODY UNDER SEVERAL PHOSPHORUS LOAD REDUCTION SCENARIOS

A) Morphometric and Hydrologic Data:

1) Volume = $4.55 \times 10^{11} m^3$

2) Surface area = $2.6 \times 10^{10} m^2$

3) Mean depth (volume/surface area) = 17.7 m

4) Hydraulic residence time (volume/annual inflow volume) = 2.6 yr.

5) Phosphorus residence time (phosphorus content/phosphorus load) = 0.56 yr.

B) Phosphorus Loading Data:

1) 1975 phosphorus load -

a) point sources[a]:	6.6×10^6 kg/yr
b) non-point sources[b]:	5.2×10^6 kg/yr
total load =	1.2×10^7 kg/yr
=	0.46 g $P/m^2/yr$

2) 1975 phosphorus load minus detergent phosphate -

a) point sources[a]:	4.3×10^6 kg/yr
b) non-point sources[b]:	5.2×10^6 kg/yr
total load =	9.5×10^6 kg/yr
=	0.37 g $P/m^2/yr$

3) 1975 phosphorus load minus 90 percent point source loading -

a) point sources[a]:	6.6×10^5 kg/yr
b) non-point sources[b]:	5.2×10^6 kg/yr
total load =	5.8×10^6 kg/yr
=	0.23 g $P/m^2/yr$

4) 1975 phosphorus load minus 90 percent point source loading minus 40 percent non-point source loading -

a) point sources[a]:	6.6×10^5 kg/yr
b) non-point sources[b]:	3.1×10^6 kg/yr
total load =	3.7×10^6 kg/yr
=	0.15 g $P/m^2/yr$

Explanation:

[a]assumed to consist solely of sewage treatment plant inputs.

[b]includes atmospheric inputs.

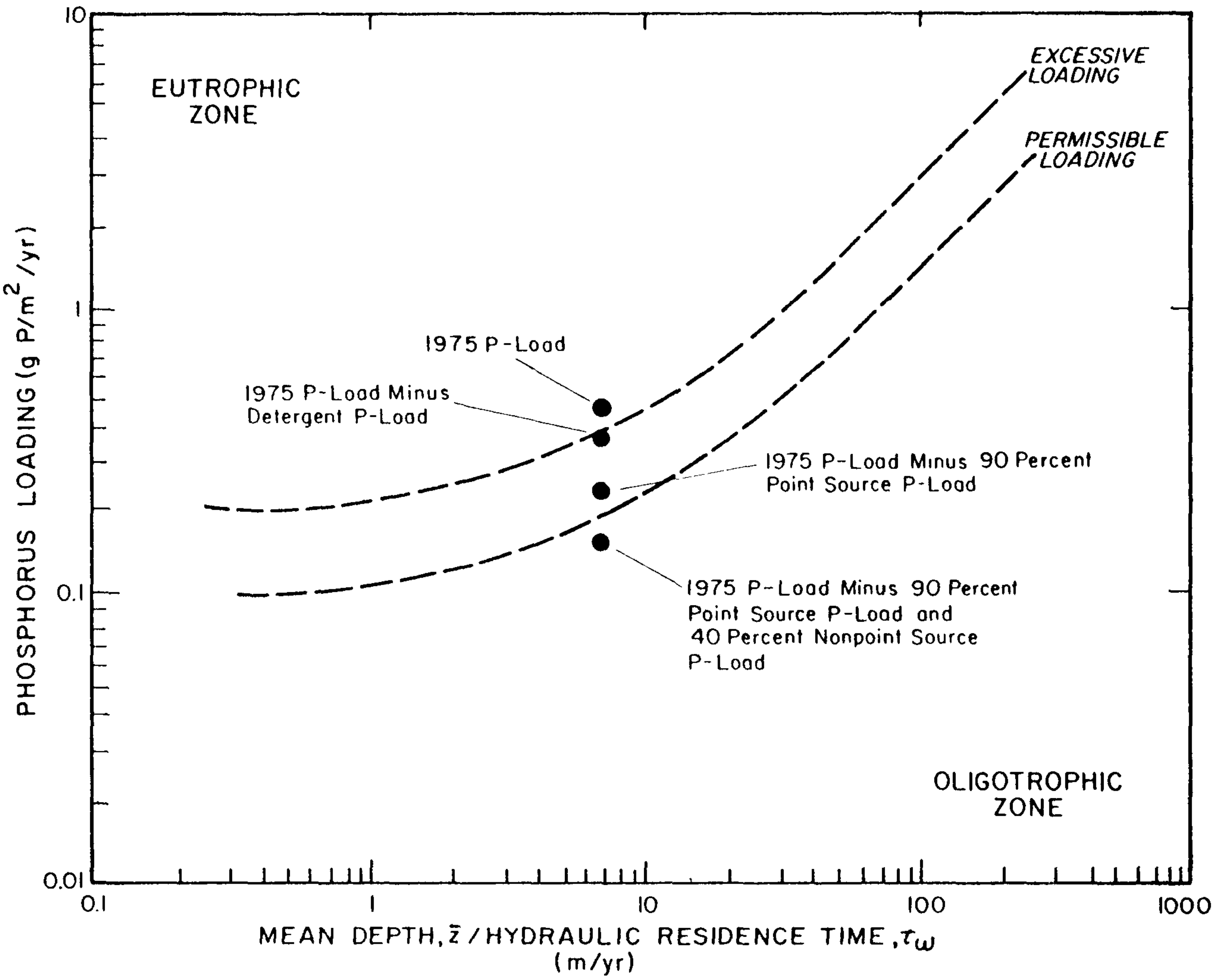

Figure 81. Phosphorus Loading and Mean Depth/Hydraulic Residence Time Relationship as Applied to Hypothetical Water Body Under Several Phosphorus Loading Scenarios

zone of the Vollenweider phosphorus loading diagram, based on the water body's mean depth and hydraulic residence time characteristics. When the detergent phosphate ban is considered, there is a discernible decrease in the phosphorus load (e.g. approximately 20 percent decrease in total phosphorus load), as indicated in Figure 81. It is important to note that Figure 80 is based on total phosphorus loadings to the hypothetical water body, which may not properly reflect the phosphorus input which is available for utilization by the phytoplankton populations in the water body. It is reasonable to suggest that the decrease in the available phosphorus fraction of the phosphorus load to the hypothetical water body will be somewhat less than shown in Figure 81.

If the point source load is reduced by 90 percent, as would be seen with advanced waste treatment phosphorus removal techniques, there is a relatively large decrease of approximately 50 percent in the total phosphorus load. This would place the hypothetical water body in the mesotrophic zone of the Vollenweider diagram, based on its mean depth and hydraulic residence time characteristics. The reduction of the phosphorus input from non-point sources can potentially be achieved by a variety of means such as control of agricultural use of fertilizers and animal manures, improved street sweeping to minimize phosphorus derived from urban drainage, and/or the control of atmospheric inputs of phosphorus. As shown in Table 28, a 90 percent point source and 40 percent nonpoint source phosphorus removal program reduces the areal phosphorus load to 0.15 g $P/m^2/yr$. The impact of advanced waste treatment for point source phosphorus removal, plus control of the diffuse sources, places the hypothetical water body in the oligotrophic zone of the Vollenweider phosphorus loading diagram (Figure 81). As discussed in another section of this report, it is important to emphasize that a change in the position of a water body, based on an altered phosphorus load, from just above, or just below, the excessive and permissible loading lines in a Vollenweider phosphorus loading diagram does not necessarily translate into a significant change in water quality. A lake may change from the eutrophic to mesotrophic zone as a result of an altered phosphorus load and still not experience a significant change in water quality.

Figures 82, 83 and 84 can be used to evaluate the expected changes in water quality resulting from various phosphorus loading reduction scenarios. In order to inject realism into the use of this model, as well as the others developed in this section, it will be assumed that the chlorophyll a concentration of the hypothetical water body does not lie exactly on the line of best fit. Changes in the water quality parameters can then be determined by moving the data point parallel to the line of best fit in the model. Table 29 and Figure 82 indicate a chloro-

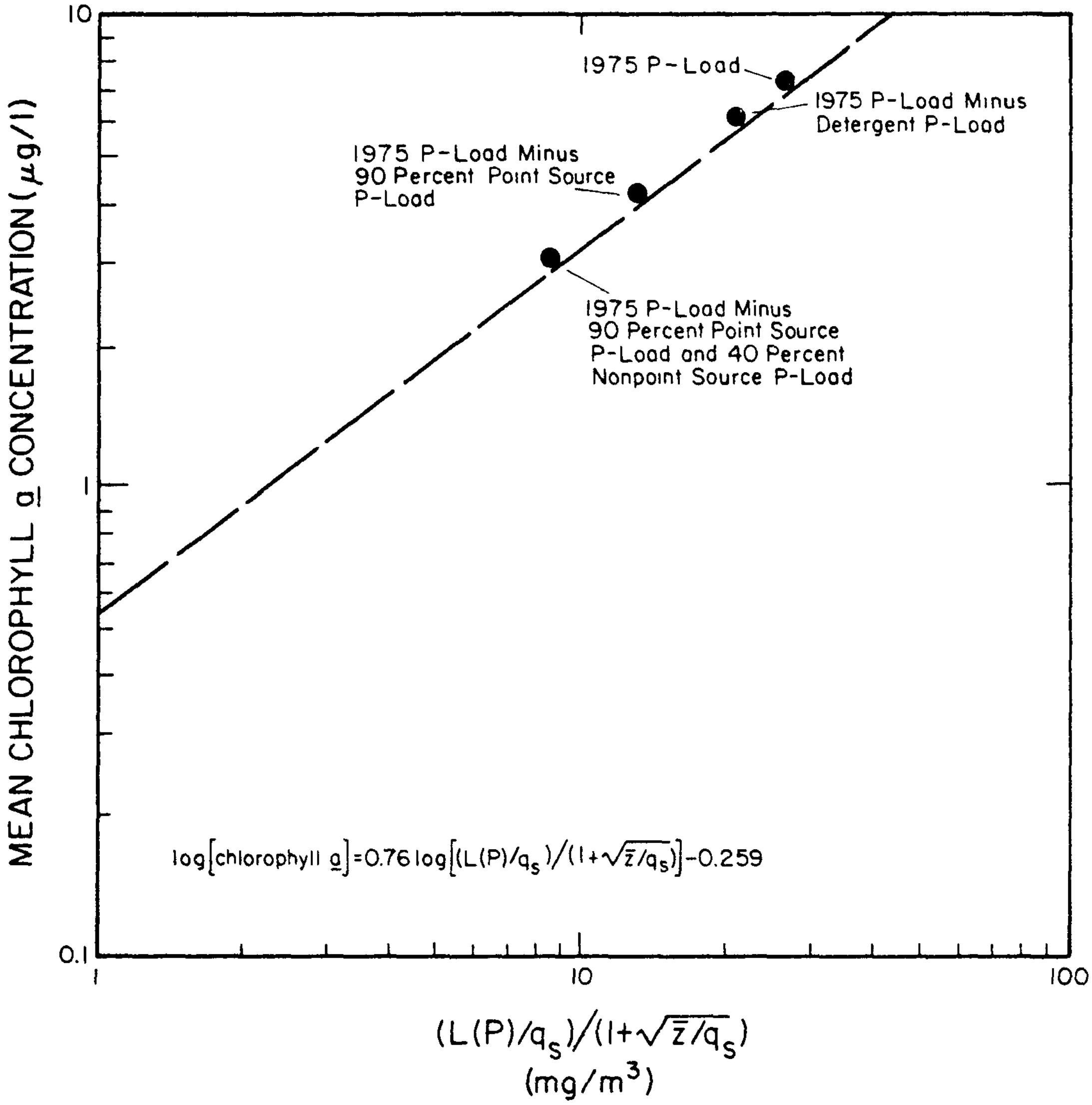

Figure 82. Phosphorus Loading Characteristics and Mean Chlorophyll a Relationship as Applied to Hypothetical Water Body Under Several Phosphorus Loading Scenarios

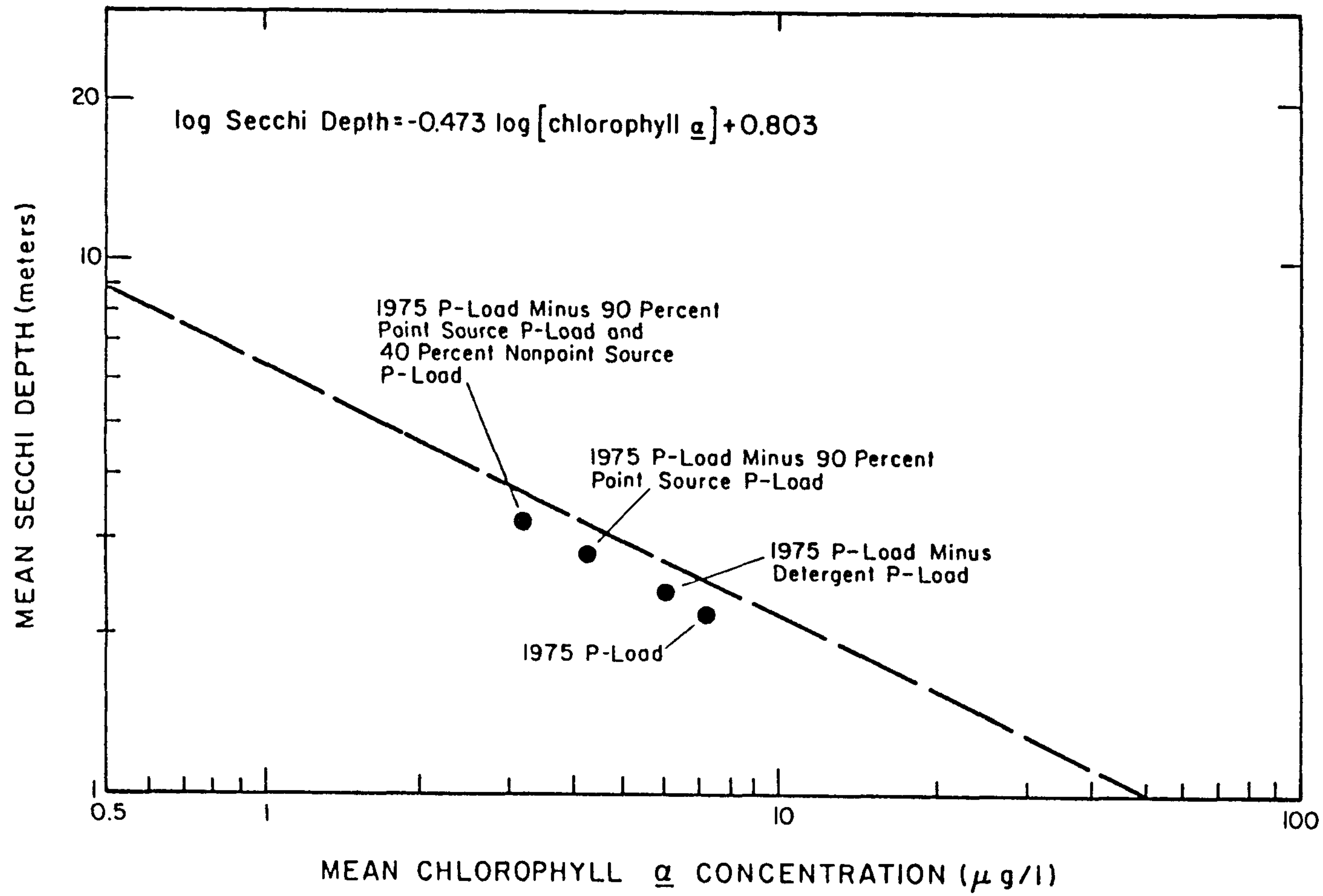

Figure 83. Secchi Depth and Mean Chlorophyll a Relationship as Applied to Hypothetical Water Body Under Several Phosphorus Loading Scenarios.

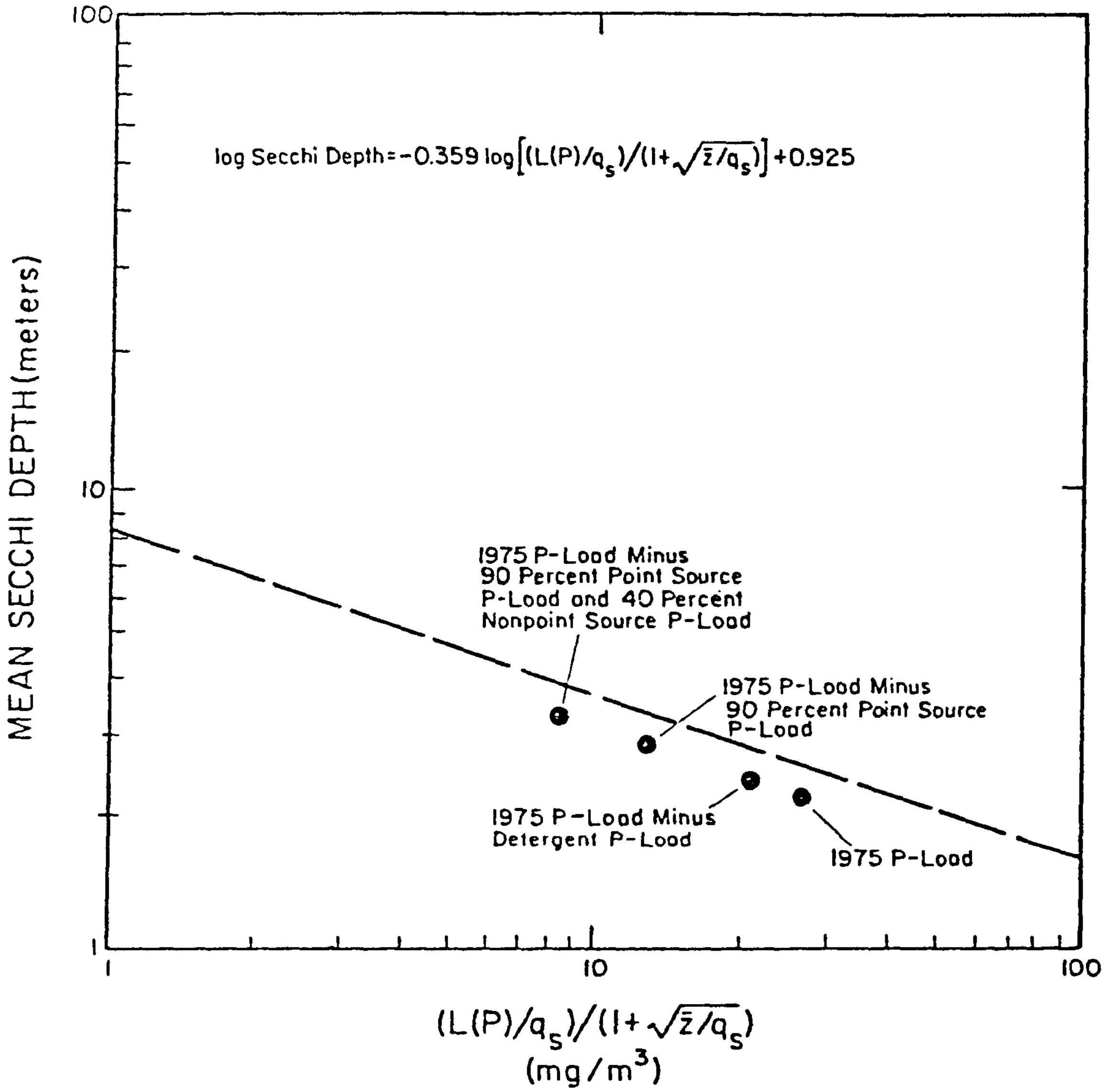

Figure 84. Phosphorus Loading Characteristics and Secchi Depth Relationship as Applied to Hypothetical Water Body Under Several Phosphorus Loading Scenarios.

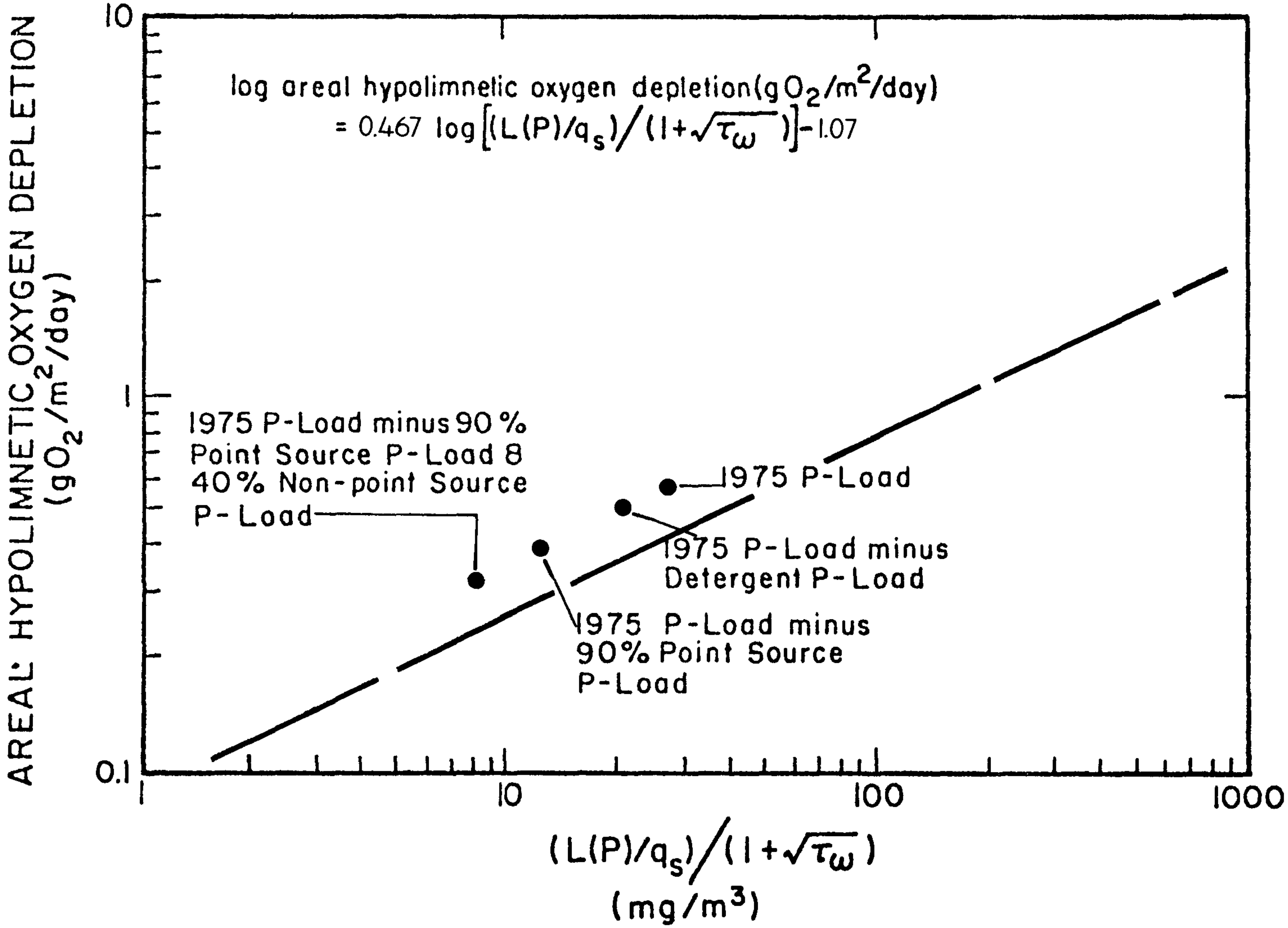

Figure 85. Phosphorus Loading Characteristics and Hypolimnetic Oxygen Depletion Relationships as Applied to Hypothetical Water Body Under Several Phosphorus Loading Scenarios

phyll a concentration of 6.5 μg/l, based on the 1975 phosphorus load. On the basis of a detergent phosphate ban alone, there will be a decrease of approximately 1.0 μg/l in the chlorophyll a concentration of the hypothetical water body. It should be noted that changes of this magnitude are frequently within the experimental error normally associated with chlorophyll a measurements on a lake-wide basis. On the other hand, a noticeable change will be seen when the 90 percent point source phosphorus removal scenario is considered. The mean chlorophyll a concentration will drop from about 6.5 μg/l to 3.9 μg/l, a decrease of approximately 40 percent. A decrease of this magnitude is significant and a noticeable increase in water quality, as reflected in chlorophyll a content, would likely result in this hypothetical water body. Finally, if a 90 percent point source and a 40 percent non-point source phosphorus reduction are considered, an additional decrease of 1.1 μg/l chlorophyll a will be seen. The chlorophyll a concentrations will have decreased from 6.5 μg/l to 2.8 μg/l. This constitutes a 57 percent decrease in the total chlorophyll a concentration in the water body compared to a 66 percent decrease in the total phosphorus load. This low chlorophyll a level is typical of unproductive water bodies, and would be consistent with the oligotrophic status of the hypothetical water body as indicated in the Vollenweider phosphorus loading diagram (Figure 81).

The changes in Secchi depth which would be expected to result from the various phosphorus load reductions are indicated in Table 29 and Figures 83 and 84. The predicted Secchi depth based on the hypothetical 1975 phosphorus load will be approximately 2.6 m. If a detergent phosphate ban is considered, the Secchi depth will increase approximately 0.2 m. This is equivalent to an increase of about 8 percent resulting from a 35 percent reduction in the point source loading. As with the chlorophyll a concentration, this amounts to an essentially undetectable change in the Secchi depth on the basis of a detergent phosphate ban alone in the hypothetical water body. The change is more significant when the 90 percent reduction in point source phosphorus loads is considered. The Secchi depth increase will be 0.8 m, a definitely discernible Secchi depth increase of about 30 percent for the 50 percent decrease in the total phosphorus load. Finally, when both the 90 percent point source and 40 percent non-point source phosphorus load reduction is considered, the Secchi depth increases from 2.6 to 3.9 m, an overall increase of 1.3 m. This constitutes a 33 percent overall increase in Secchi depth for a 66 percent overall decrease in phosphorus load.

The hypolimnetic oxygen depletion rate changes to be expected from the various phosphorus loading scenarios is indicated in Figure 85. The predicted 1975 areal hypolimnetic oxygen depletion rate is 0.60 g O_2/m^2/day. When the detergent phosphate ban is considered, the rate decreases approximately 0.1 g O_2 for each m^2 of hypolimnetic area. This is a 17 percent decrease for a 33 percent decrease in the point source phosphorus loading. The 90 percent point source phosphorus loading reduction results are more significant, with the hypolimnetic oxygen depletion rate dropping to 0.37 g O_2/m^2/day. This corresponds to a 38 percent decrease in the oxygen depletion rate for a 50 percent reduction in the phosphorus load. When both the point and non-point phosphorus load reductions are considered, the hypolimnetic oxygen depletion rate decreases to 0.32 g O_2/m^2/day, an overall decrease of 47 percent for an overall 66 percent reduction in the phosphorus load to the hypothetical water body.

The improved water quality associated with 40 percent control of phosphorus from diffuse sources will almost certainly be less than that predicted in Table 29. As a result of the fact that many diffuse sources of phosphorus such as urban and rural drainage and the atmosphere usually have large parts of their phosphorus in a particulate form, much of which is unavailable to support algal growth.

Several points should be noted on the use of this approach. First, it is important to emphasize that the magnitude of the changes discussed in the chlorophyll *a* or Secchi depth relationships refer to changes associated with planktonic algal growth. At present, there is no information available for reliably predicting the effects of a reduced phosphorus load on the growth of *Cladophora* and other attached algae, as well as the growth of macrophytes and floating macrophytes, such as water hyacinths and duckweed. There is also no information available for reliably predicting the effects of a phosphorus load reduction on water clarity in the nearshore waters of a water body. Further, water bodies will not adjust immediately to an altered phosphorus load. Rather, it will require a period of time equal to three

Table 29. SUMMARY OF PHOSPHORUS LOADING CHARACTERISTICS, CHLOROPHYLL a AND SECCHI DEPTH OF HYPOTHETICAL WATER BODY UNDER SEVERAL PHOSPHORUS LOAD REDUCTION SCENARIOS

Phosphorus Loading Situation[a]	$(L(P)/q_s)/(1+\sqrt{\bar{z}/q_s})$ (mg/m^3)	Chlorophyll a (μg/l)[b]	Secchi Depth (m)[c]	Hypolimnetic Oxygen Depletion (g O_2/m^2/day)
1975 Phosphorus Loading	25.9	6.5	2.6	0.60
1975 Phosphorus Loading Minus Detergent Phosphate	20.8	5.5	2.8	0.50
1975 Phosphorus Loading Minus 90% Point Source Loading	13.0	3.9	3.4	0.37
1975 Phosphorus Loading Minus 90% Point Source and 40% Non-Point Source Loading	8.4	2.8	3.9	0.32

[a]Phosphorus loadings were taken from Table 28.

[b]As determined in Figure 83, based on phosphorus load characteristics indicated in this table.

[c]As determined in Figure 84, based on phosphorus load characteristics indicated in this table.

[d]As determined in Figure 85, based on phosphorus load characteristics indicated in this table.

phosphorus residence times (Sonzogni *et al.*, 1976) before a new equilibrium condition will be established in the water body. The models presented in Figures 22, 79, and 80, may be applicable when a new equilibrium state is reached in a water body. For example, the use of this approach predicts a chlorophyll *a* concentration in Lake Ontario of about 4.5 µg/l, based on its 1973 phosphorus load. However, chlorophyll *a* values reported by Dobson (1975), the International Joint Commission (1976b) and Vollenweider (1976a) are in the order of six to eight µg/l for the openwaters of Lake Ontario. These higher values are possibly due to a non-equilibrium condition of Lake Ontario resulting from its reduced phosphorus load. Lake Ontario has not yet had sufficient time to respond to this reduced phosphorus load. However, the use of these models in successfully predicting chlorophyll *a* concentrations and Secchi depths in Lake Washington following completion of its sewage diversion project, which was discussed earlier in the section, lends considerable support to these approaches in assessing the resulting water quality in a water body following a change in its phosphorus load. When this approach is applied to the Great Lakes, the results obtained with the use of Figures 22, 78, 79, and 80 are in general agreement with the observations of Gilbertson *et al.* (1972), Vollenweider *et al.* (1974) and Dobson (1975) concerning the Great Lakes.

In the Great Lakes consideration has to be given to the fact that the nearshore waters of the lakes often have elevated concentrations of nutrients compared to the open water. This situation arises from the strong longshore currents which tend to be present in large water bodies and which inhibit mixing of nearshore with offshore water. Under these conditions, a different mean depth/hydraulic residence time relationship should be used in order to predict nutrient load-response relationships than would be applicable to the open waters of the lakes.

Results of computations such as those presented above on nutrient load-response relationships provide water quality managers and the public with the information needed to evaluate the magnitude of water quality improvement associated with a particular nutrient control strategy. In order to develop a meaningful nutrient control program it is necessary to evaluate the costs associated with each approach. The cost of each nutrient control program and the degree of water quality improvement can be used to choose the most cost-effective control program. Prior to the development of these relationships between phosphorus loading and water quality, as measured by chlorophyll a concentrations, Secchi depth, and hypolimnetic oxygen depletion, there was no readily available, and reliable, method for predicting the expected improvement in water quality resulting from a reduction in the phosphorus loading to a water body by a certain degree. Water quality managers can now develop cost-effective eutrophication control programs in which they can inform the public of the degree of improvement in water quality expected to result from expenditure of funds by a certain amount. The taxpayers can then decide how much they are willing to pay for improved water quality.

APPLICATION OF RESULTS TO IMPLEMENTATION OF SECTION 314-A OF PUBLIC LAW 92-500

Section 314-A of PL 92-500 requires that each state classify its lakes and impoundments with respect to their degree of fertility. Furthermore, the states must develop a nutrient control program to minimize fertility in those water bodies found to be excessively fertile. The results of this investigation of the US OECD eutrophication study provide the states and the federal government both with a basis by which this type of classification can be made, and with the ability to assess the improvement in water quality that is likely to result from a nutrient control effort of a certain magnitude. From a water quality management point of view, the expected improvement from a nutrient control program can be weighed against the cost of achieving the nutrient control, and a decision can be made as to whether the control effort will result in a sufficient improvement in water quality to justify the expense of the program. It is important to look on the results of this study as a guide for implementation of public policy in the area of excessive eutrophication of natural waters. While Vollenweider, and others who have modified his approach, have been able to formulate nutrient load-eutrophication response relationships with a relatively simple methodology, involving normalizing water bodies based on their mean depth and hydraulic residence time, there are many other factors which can influence the nutrient load-algal response relationship in natural waters.

Examination of the various plots presented in this report show there is considerable scatter in the data. Part of this data scatter is due to differences in measurement techniques. Another part is due to the inherent variability of lakes and impoundments. With respect to measurement, every point on any of the nutrient load-response diagrams usually has considerable variance in both the x and y directions. One of the most difficult parameters to estimate for many water bodies is the hydraulic residence time. This factor is continuously changing. A series of wet years could markedly affect the results compared to more normal or dry conditions. For example, an impoundment in north central Texas shows a hydraulic residence time from 0.3 years to 22 years, with a mean of about 4 years, dependent upon wet and dry climatic cycles. In addition to variable climatic patterns, another factor to be considered is that of short-circuiting of the inflow and outflow waters, such that the inflowing nutrients do not interact with the total water body. This may be an especially significant problem for large, deep impoundments. Under these conditions, a modification of the hydraulic residence time term should be made to more properly reflect the actual behavior of the nutrients in the impoundment. This modification should reflect the fact that some of the nutrients that enter a water body may leave it by way of outflow before they have had the opportunity to interact with the phytoplankton.

The variance about the vertical displacement on the diagrams in this report, for a phosphorus load, chlorophyll, Secchi depth or hypolimnetic oxygen depletion response, is likely to be very large under certain circumstances. The data presented in this report are often based on a single year's measurements. Lakes and impoundments respond to nutrients not only for the year in which the nutrients are added, but for previous years' nutrient inputs as well. Each water body would have an individual response in this respect. Further work, which will not be reported in this report, is being done by these authors to estimate the magnitude of the associated variance that is likely to be encountered with measurements of the various load-response relationships. From the work done thus far, it is important that no regulatory agency require implementation of a control program because a water body's phosphorus load plots just above the permissible or excessive line on the diagrams presented in this report. Similarly, no regulatory agency, or other group, should assume that because a lake plots just below the excessive or permissible line, that this lake will not have water quality problems due to excessive fertilization.

Factors such as color, turbidity, morphological shape of the water body basin, rainfall runoff patterns, characteristics of the watershed, etc., all would have an influence on the nutrient load-response relationships in natural waters, and all contribute to the scatter of points in the various nutrient load-response evaluations made in this study. One of the areas of research that needs considerable additional attention, in attempting to reduce the scatter in the data, is that of available nutrients. In general, the various diagrams presented in this report are based on total phosphorus. It is well known that only part of this total phosphorus is available. As a guideline, Cowen and Lee (1976b) have concluded that the best approach for estimating the available phosphorus load to a water body is that it is equal to the soluble orthophosphate load plus 20 percent of the difference between the total phosphorus and soluble orthophosphate load. This difference between the total and soluble orthophosphate is made up of inorganic and organic forms which are generally particulate. These results indicate that for those water bodies in which the primary source of nutrients is from agricultural or land runoff, a substantial part of the phosphorus may not contribute to the algal-available load.

Another factor to consider concerns the amounts of available phosphorus that reach a water body when the origin of the phosphorus load is a considerable distance from the water body. For example, in the US-Canadian Great Lakes, some controversy has developed concerning the significance of domestic wastewater discharges many miles from the lake in influencing excessive fertilization problems in the Great Lakes. If the wastewaters enter a lake somewhere between their origin and the Great Lakes, most of the nutrients would be retained in the intermediate lake, since many water bodies trap from 60 to 90 percent of the phosphorus that enters them by incorporation of the phosphorus into the sediments. Further, as the available phosphorus added to a stream some distance from the lakes mixes with the erosional materials, and/or is utilized in various biological processes, it is becoming less and less available for stimulation of algal growths. It is likely that available nutrients discharged to rivers which are considerable distances from the lake of interest will have much less influence on stimulating extensive fertilization problems than would the same nutrients discharged directly to the water body.

Special consideration in assessing nutrient sources should be given to septic tank wastewater disposal systems since a large part of the US population utilizes this form of wastewater disposal. A comprehensive review on the significance of septic wastewater disposal systems as a source of phosphorus has recently been completed by Jones and Lee (1977). They concluded that, with few exceptions, the phosphorus present in septic tank domestic wastewater disposal systems will have little influence on stimulating excessive fertilization problems in natural waters. Guidance is provided by Jones and Lee in evaluating, on a case by case basis, whether in a localized area excessive fertilization problems are caused by septic tank systems.

While this report has focused primarily on the application of the Vollenweider loading approach to assessing water quality in which the water quality problems are related to excessive fertilization for whole bodies of water, it is applicable to parts of a water body as well. A number of the lakes and impoundments investigated in this report were subdivided into various sections or arms. The results appear to indicate that this approach is appropriate. Further, for large water bodies such as the Great Lakes, the approach described in this report should be applicable to bays and nearshore waters as well. In the case of the Great Lakes, it is important to be able to estimate the exchange of water between the nearshore and offshore zones, or between the openwaters of the lakes and their bays. It is important to emphasize once again that the Vollenweider loading approach is directly applicable to the management of water quality problems associated with excessive fertility, as manifested in phytoplankton growth which cause an impairment of recreational use of water. Further research is likely to produce the information needed to develop modifications of the Vollenweider loading relationship for other water quality problems such as excessive growth of macrophytes, attached algae, dissolved oxygen depletion in the hypolimnion and impairment of water supplies for domestic and industrial use. Further work, which will be reported by these authors in subsequent reports, is being done along these lines in order to define conditions for which the Vollenweider loading approach is not applicable. It is already apparent from this study that the Vollenweider approach must be modified for those water bodies which show very short hydraulic residence times because the nutrients entering into the water body could pass through it before interacting with the phytoplankton and thus would not produce an algal crop in the water body proportional to its nutrient loading. In these cases, the Vollenweider loading approach, in its present form, would be inappropriate for assessing the eutrophication status of the water body.

AN APPROACH FOR THE USE OF THE VOLLENWEIDER NUTRIENT LOAD-WATER QUALITY PROGRAM

The procedures that should be utilized in applying the Vollenweider loading relationship for the development of a water quality management program designed to improve water quality or minimize future deterioration are presented below.

1. Determine the limiting nutrient. Since the Vollenweider loading relationship was derived for phosphorus, the first step in its application would be to determine whether phosphorus or nitrogen is the limiting nutrient in the water body. This assumes that all other factors affecting algal growth (i.e., light and temperature) do not limit the maximum algal biomass that will develop and that it is the concentration of the limiting nutrient, relative to the stochiometric needs of the algae, which controls or limits the deterioration of water quality.

The limiting nutrient can usually be determined by several techniques, including N:P ratios, bioassay studies or simple observation of the available nutrient concentration dynamics over the seasonal and/or annual cycle (Lee, 1973). The use of the growing season inorganic nitrogen: soluble orthophosphate mass ratio (expressed as N:P) in a water body was discussed in an earlier section of this report (see Tables 9 and 10). Bioassay techniques can also be used to determine the limiting nutrient in a water body. The algal assay procedure provides a standardized test for identifying algal-growth-limiting nutrients in water bodies, for determining the biological availability of algal growth-limiting nutrients, and for quantifying algal responses to changes in concentrations of the nutrients (Sridharan and Lee, 1977). An estimate of the limiting nutrient can be obtained by observing the dynamics of the available nutrients during the growing season. If one of the algal-available nutrient forms becomes depleted in a water body at the same time that the other is still present in large quantities, it is usually reasonable to assume that the depleted nutrient may be the algal-growth-limiting nutrient.

If it is determined that nitrogen, rather than phosphorus, is the aquatic plant growth-limiting nutrient, then two options are available. One can either attempt to control the nitrogen loading, or else reduce the phosphorus loadings to such an extent that phosphorus becomes the limiting nutrient. The latter course of action is almost always preferred, for reasons mentioned in earlier sections of this report (Vollenweider, 1968; 1975a; Lee, 1971; 1973; Vallentyne, 1974; Golterman, 1976). It does not matter that nitrogen initially controls the algal growth in a water body, but rather that phosphorus can be made limiting in the water body. To determine if it is possible to change a water body from nitrogen-limitation to phosphorus-limitation, one must be able to assess the potential benefit that might be derived from a reduction of phosphorus in a water body. Sridharan and Lee (1977) have recently developed a technique for making such an assessment. This procedure is based on studying the response of algae to alum-treated lake water and has worked well for evaluation of the potential benefit to be derived from a phosphorus reduction in Lake Ontario. Based on the results of these types of analyses, one can determine the limiting nutrient in a water body, and make an evaluation of the potential benefits, in terms of algal growth responses, to be derived from a decrease in the phosphorus content of a water body.

2. Determine the available nutrient sources and significance of each source. This step consists of quantifying the nutrient loadings to a water body. It is first necessary to identify all the sources of nutrient inputs, both point and non-point sources. Sonzogni and Lee (1974) have presented an extensive examination of the estimated nutrient loadings to Lake Mendota in 1972. The approach used by Sonzogni and Lee is an example of how one may assess the nutrient sources to a water body. They examined the nutrient inputs from waste water discharges, urban, rural and forest runoff, groundwater seepage, baseflow, nitrogen fixation and from the atmosphere directly onto the lake surface. They then determined the total nutrient loadings from these sources. This same approach can be used to assess the nutrient sources for most water bodies.

Once the sources are identified, one may then quantify the nutrient inputs for a water body. The loadings may be directly measured or determined by indirect methods. If it is measured directly, the sampling program should be sufficient to allow one to determine the variability from a particular source. For example, the amount of phosphorus from a sewage treatment plant can be determined by measuring the phosphorus concentration in the effluent and multiplying this concentration by the flow. The result will be the mass of phosphorus loading from this source. However, the phosphorus concentrations in sewage treatment plants can vary widely over a daily, weekly and monthly cycle. This variability must be determined so that accurate loads from this major nutrient source can be computed. Another case of variability involves measurement of land runoff. According to Kluesner and Lee (1974), the phosphorus concentration in urban runoff after a storm varies widely, usually reaching a peak which is not coincident with the peak runoff flow. Thus, the concentrations and flows may have to be measured frequently during a storm if it is desirable to get very accurate loading estimates during this period.

An alternative to direct measurement is to use watershed land use nutrient export coefficients. This method was used in this report and is described in detail in an earlier section. This method is based on the fact that a given land use activity within a watershed will produce a relatively constant nutrient export over an annual cycle (i.e., an acre of corn field or urban area will produce about the same annual export of phosphorus and nitrogen). Thus, loadings to a given water body can be determined on the basis of land use type in the water body's watershed and use of the appropriate nutrient export coefficient. A number of studies concerning export coefficients for various land uses have recently been completed (Vollenweider, 1968; Sonzogni and Lee, 1974; US EPA, 1974c; Vollenweider and Dillon, 1974; Uttormark et al. 1974; Dillon and Kirchner, 1975; Dillon and Rigler, 1975). One can determine phosphorus and nitrogen loadings from sewage treatment plants in a similar manner. Several studies have been conducted to estimate per capita nutrient concentrations in domestic wastewaters (Vollenweider, 1968; Sonzogni and Lee, 1974; Dillon and Rigler, 1975). One may use these reported values or experimentally determine the per capita loadings by direct measurements. The reader is referred to these various studies for appropriate nutrient export coefficients on per capita inputs. If it is felt that a given export coefficient is not accurate for a given land use, an alternative is to directly measure the nutrient runoff from a land use type in the watershed and formulate one's own coefficients.

Using these methods, the loadings of total phosphorus and nitrogen, as well as algal-available phosphorus and nitrogen to a water body can be computed. One can also then evaluate the relative significance of each source if it is necessary to choose between controlling the input from several sources. The nutrient loadings from domestic sewage treatment plants is usually one of the most significant sources for most water bodies. One should also evaluate the loadings of available nutrients versus the loadings of total phosphorus and nitrogen, since the available nutrients are the ones assimilated by algal populations in water bodies. As mentioned earlier (Cowen and Lee, 1976b), the best estimate of the loading of available phosphorus is that it is equal to the sum of the available phosphorus loading plus 20 percent of the difference between the total phosphorus and available phosphorus loading.

3. Assess the nutrient load-eutrophication response relationships. When an estimate of the available phosphorus loading is available, the next step is to assess the relationship between the loading and the eutrophication responses of a water body. This assessment assumes that the computed phosphorus loading is accurate. The accuracy of the phosphorus loading estimate, whether measured or computed using nutrient export coefficients, can be checked using the relationship developed by Vollenweider between the ratio of the mean total phosphorus concentration to the influent phosphorus concentration and the hydraulic residence time (see Equation 26 and Figure 14). This approach was presented in an earlier section of this report. The phosphorus and nitrogen loading estimates, if they were directly measured, could also be checked using appropriate watershed nutrient export coefficients.

After the reasonableness of the loading estimates, particularly phosphorus, has been determined, the relationships presented in earlier sections of this report can be used to assess the relative degree of oligotrophy or eutrophy of a water body. The critical phosphorus loading levels can be determined for a water body. Also, the expected enhancement or deterioration of water quality following a phosphorus loading reduction or increase, respectively, can be evaluated. This can be done in a manner completely analagous to that presented by Lee (1976) concerning the expected effects of a phosphate detergent ban in the State of Michigan on water quality in the Great Lakes.

The relative trophic condition of the water body can be determined using the Vollenweider phosphorus loading and mean depth/hydraulic residence time relationship (Figure 19). To evaluate the water quality, the relationship between phosphorus loading and mean epilimnetic chlorophyll _a_ concentration in a water body can be determined with the use of Figure 22. One can next determine the expected water clarity for a given phosphorus load with the use of Figure 79. If a large quantity of data is available for a water body concerning its chlorophyll _a_ concentrations and corresponding Secchi depths, one can construct a Secchi depth and chlorophyll _a_ concentration diagram specific for that water body. This can be transformed into a phosphorus load characteristics and Secchi depth diagram in the same manner as was done with Figure 79. Otherwise, Figure 79 can be used in its present form.

One could also use these relationships to evaluate the expected changes in water quality in a water body in future years as a function of future changes in phosphorus loads. Figures 19, 22,79 and 80 can be used in the same manner as indicated above for evaluating expected future phosphorus loads. Particularly, Figure 19 can indicate the expected relative changes in trophic condition resulting from an altered phosphorus load. Figures 22, 79 and 80 can be used to predict the expected changes in mean chlorophyll a concentrations, Secchi depth and hypolimnetic oxygen depletion for an altered phosphorus load.

4. Evaluate cost-benefit analysis of eutrophication control program. Most eutrophication control programs are based on reduction of phosphorus loads to a water body. As indicated above, the expected water quality changes can be evaluated for a given phosphorus load reduction. The final question then involves the cost-benefit of any given eutrophication control program. Previously, eutrophication control programs based on phosphorus load reduction were largely subjective in nature. The use of the above-mentioned relationships provides individuals concerned with water quality management with a quantitative tool to evaluate expected changes in water quality resulting from eutrophication control programs based on reduction of phosphorus inputs. The final question to be answered concerns evaluation of the relative monetary worth of such a eutrophication control program. Do the results of a phosphorus removal or sewage diversion program, for example, justify the funds expended for the project? In short, is the final expected product worth the money?

This final question brings social, economic and political considerations into the overall picture. Lee (1971; 1973) and Vollenweider and Dillon (1974) have determined that widespread use of phosphorus removal programs is economically feasible. Lee (1971; 1973) has determined that phosphorus removal from domestic wastewaters is possible for a cost of about one cent per person per day. It is then up to those individuals concerned with water quality management to determine if it is worth one cent per person per day to produce a change in water quality as predicted with the use of Figures 22, 79 and 80 For example, if it is shown that the phosphorus loading to a water body can be reduced by 60 percent by initiating advanced waste treatment for phosphorus removal from domestic waste waters, and that such a reduction will lower the mean chlorophyll a concentration from 10 μg/l to 5 μg/l and raise the Secchi depth from 1 meter to 2 meters, is the cost of building and operating the plant justified by the expected improvements in water quality? This will have to be evaluated on an economic and political level since such programs are usually ultimately funded by the taxpayers. The important point to be made is that now the individuals who must pay for eutrophication control programs can be shown in advance of the initiation of such programs what they will get in terms of improved water quality for their money. They can then decide, by whatever means they choose, whether the expected improvements are worth the expected costs to them.

It is expected by these authors that additional quantitative tools for evaluating predicted changes in water quality will be developed in the future, providing further methodologies for making water quality management cost-benefit analysis decisions.

SECTION XII

TROPHIC STATUS INDEX STUDY

The US OECD data base offers an opportunity to examine the comparability and to some degree the reliability of several recently-proposed water body trophic status indices. This section of this report is devoted to a review of these trophic status index schemes and an analysis of the results of the trophic classifications of the US OECD water bodies.

GENERAL CONSIDERATIONS

Lakes and other surface waters are characteristically divided into two general categories, oligotrophic and eutrophic. Further, it is generally agreed that mesotrophic describes water bodies in a transition state between oligotrophic and eutrophic (Fruh et al. 1966; Vollenweider, 1968; Lee, 1971; Vallentyne, 1974). However, the exact meaning of these three terms is still debated among limnologists because of a lack of understanding concerning details of the eutrophication process, other than on a gross level, and its effects on the aquatic environment.

Weber (1907, as cited in Hutchinson, 1969), was the first to introduce the terms "eutrophic" and "oligotrophic." He used these terms to describe the general nutrient conditions of soils in German bogs. The succession of Weber's scheme ran from eutrophic to oligotrophic as a submerged bog was built up to a raised bog. The submerged bog was characterized as eutrophic or well-nourished, while the raised bog was characterized as oligotrophic. Naumann (1919, as cited in Hutchinson, 1969), introduced these terms into limnology. Naumann used the term "eutrophic formation" to describe a phytoplankton assemblage in nutrient-rich waters. Naumann (1931, as cited in Stewart and Rohlich, 1967) later refined his definition of eutrophication as "an increase of the nutritional standards (of a body of water), especially with respect to nitrogen and phosphorus."

As originally defined, eutrophic and oligotrophic referred to water types (i.e., quality of water). However, the term has generally come to refer to general lake types, including the physical, chemical and biological characteristics of the water body and its drainage basin (Brezonik et al., 1969). The difficulty in defining the terms oligotrophic and eutrophic is related to the fact that these terms are used in different ways by different investigators. Some use these terms to refer to aquatic plant nutrient flux, others use them to describe plant and animal production, while even others

Table 30. GENERAL CHARACTERISTICS FREQUENTLY USED TO CLASSIFY WATER BODIES

Parameter	General Characteristic: Oligotrophic	Eutrophic
Aquatic plant production	low	high
Algal blooms	rare	many
Algal species variety	many	variable to few
Characteristic algal groups	---	blue-green *Anabena* *Aphanizomenon* *Microcystis* *Oscillatoria rubescens*
Littoral zone aquatic plant growth	sparce	abundant
Aquatic animal production	low	high
Characteristic zooplankton	*Bosmina obturirostris* *B. coregoni* *Diaptomus gracilus*	*B. longirostris* *D. cucullata*
Characteristic bottom fauna	Tanytarsus	Chironomids
Characteristic fish	deep-dwelling, cold water fishes such as trout, salmon and cisco	surface-dwelling, warm water fish such as pike, perch and bass; also bottom dwelling fish such as catfish
Oxygen in the hypolimnion	present	absent
Depth	tend to be deeper	tend to be shallower
Water quality for most domestic and industrial use	good	poor
Total salts or conductance	usually lower	sometimes higher
Number of plant and animal species	many	few

Taken from Fruh *et al.* (1966); Lee (1971); Vallentyne (1974) use them to describe the process of excessive discharge of aquatic plant nutrients to a water body that results in water quality deteriotation (Lee, 1971).

Even though there is general agreement concerning a oligotrophic-eutrophic succession scheme, the problem of the trophic status or classification of a water body at a given point in time remains to be considered. This illustrates a basic problem in lake classification, namely that the exact classification of a water body is usually related to its intended use. A water supply reservoir manager would likely have a much more stringent definition of eutrophic than would a fisherman who was interested in fish production. They would desire opposite ends of the trophic spectrum; hence, their views of oligotrophy versus eutrophy would also likely be different.

However, there are some relatively widely-accepted general characteristics used to characterize lakes. Table 30 summarizes the commonly accepted characteristics of oligotrophic and eutrophic lakes. The reader is also referred to recent reviews of the eutrophication process and its manifestations (Sawyer, 1966; American Water Works Association, 1966; Fruh et al., 1966; Stewart and Rohlich, 1967; Vollenweider, 1968; Brezonik et al., 1969; Federal Water Quality Administration, 1969; National Academy of Sciences, 1969; Lee, 1971; Likens, 1972a; US EPA, 1973a; and Vallentyne, 1974).

Examination of Table 30 shows that oligotrophic lakes tend to have a low nutrient flux relative to their volume of water. They contain small amounts of organisms, but many different species of both aquatic plants and animals. In general, oligotrophic lakes are deep, with average depths of 15 meters or greater and maximum depths frequently greater than 25 meters (Vallentyne, 1974). However, this feature is highly variable. Further, as oligotrophic lakes fill, due to sediment deposition over time, they will tend to become eutrophic (Lee, 1971). Oligotrophic lakes usually have high dissolved oxygen concentrations in the hypolimnion during all periods of the year, including the growing season. This oxygen-containing, cool hypolimnetic region is the home of the trout, walleye, cisco and other cold water prized game fish sought by fishermen. The water quality is generally good the year round in oligotrophic lakes. In general, oligotrophic lakes can be characterized as deep, transparent water bodies with a low nutrient flux relative to their ability to assimilate the nutrients.

By contrast, eutrophic water bodies have a high nutrient flux relative to their water volume. As a result, they are highly productive water bodies with large amounts of aquatic life, but of somewhat fewer species than oligotrophic lakes. They are highly productive at all trophic levels and frequently experience algal blooms, especially during the growing season. Characteristic algae include the nuisance blue-green species associated with deteriorated water quality. The same is true for all other aquatic life species. The fish are usually the "coarser" species not generally sought by most fishermen (though this may vary from location to location). They are generally shallow, often with extensive littoral areas with abundant plant growths. Mats of macrophytes and attached algae may carpet the littoral zone, depending on competition between planktonic and attached plants and on the normally higher turbidity waters in eutrophic water bodies. Eutrophic lakes deep enough to develop a thermocline usually show a partial or complete dissolved oxygen depletion in the hypolimnion. The extent of oxygen depletion will depend on the amounts of aquatic plants that develop in the surface waters, and may become super-saturated with oxygen due to the increased photosynthetic activity at the surface. The surface water typically becomes turbid in the summer as a result of algal growth, to the extent that, with few exceptions, the amount of aquatic plants produced will be restricted to the surface waters. Such turbidity restricts light penetration to the epilimnetic waters, with the result that the Secchi depth is usually three meters or less (in contrast to the 10+ meters of some oligotrophic waters).

There also appears to be a correlation between the total dissolved salt content or conductivity and the increased primary productivity, presumably because the higher salt content is related to a high aquatic plant nutrient flux. In general, the overall water quality is poor as a result of the increased nutrient flux and resultant increased growth at all trophic levels.

In spite of these generally-accepted characteristics of productive versus nonproductive, the problem of the absolute classification of the trophic conditions of water bodies is still unsettled. Individuals tend to subjectively classify water bodies on the basis of some of the common, though arbitrary, trophic state indicators listed in Table 30 (i.e., nutrient concentrations, Secchi depth, hypolimnetic oxygen depletion, chlorophyll concentrations, etc.). A strict agreement is missing on what standards or values of these and other parameters constitutes a given trophic state. This interpretation still varies widely among investigators.

REQUIREMENTS FOR A TROPHIC STATUS CLASSIFICATION INDEX

The traditional water body classification scheme of oligotrophic, mesotrophic and eutrophic is inadequate for descriptive purposes other than in a very broad sense (Shapiro, 1975b). As a result, there has been a development of several trophic classification systems in an attempt to classify lakes on a quantitative basis. A variety of characteristics of water bodies have been used as a basis for various classification schemes and many of the schemes use markedly different approaches.

An adequate trophic index scale or scheme is particularly needed in view of the mandates of Public Law 92-500. Section 314-A of this law requires that each state classify its lakes according to their trophic condition. Further, eutrophication control measures must be initiated by the states in water bodies deemed to be excessively fertile. The question then arises as to which classification or index scheme to use. The array of trophic indices used in the past is both wide and diverse. These range from determining recreational potential, management purposes and scientific studies. The indices may be descriptive or analytical, subjective or objective, simple or complex, relative or absolute, biological, physical and/or chemical, etc. (Shapiro, 1975b).

There is need for a numerical trophic state index which will permit a more appropriate assignment of the trophic condition of a water body than the previous broad descriptions of oligotrophic, mesotrophic and eutrophic. The index scheme should be simple, based on practical parameters whose values can be determined relatively easily and which do not require sophisticated methods of statistical analysis. More complicated trophic classification indices could be developed, but would likely have limited use.

An example of a more complex scheme is a trophic state index based on the simultaneous use of multiple factors developed by Shannon and Brezonik (1972). They based their multivariate approach on seven trophic state indicators, including primary productivity, chlorophyll a, total phosphorus, total organic nitrogen, Secchi depth, specific conductivity and Pearsall's cation ratio (i.e., (Na+K)/(Ca+Mg)). Shannon and Brezonik applied their classification system to 55 lakes in Florida and found a good correlation between the trophic status index values obtained using their approach and the traditional trophic classification of these water bodies. This index has problems based on the amount of data needed for its use. For many water bodies, it is not always possible to obtain all the data needed for classification. Shapiro has criticized the Shannon and Brezonik approach since it tends to misclassify water bodies. According to Shapiro (1975b), when Shannon and Brezonik (1972) applied their trophic index system to Lake Alice, one of the 55 Florida lakes in their study, it had a TSI value of 10.7. This places it in a hypereutrophic category, relative to the other lakes in their study. However, Lake Alice has a low primary productivity and chlorophyll concentration, inconsistent with a hypereutrophic water body.

Three trophic index schemes which show varying degrees of promise have recently been developed. These include the trophic classifications of the US EPA (1974d), Carlson (1974) and Piwoni and Lee (1975). In addition, a trophic index system based on Vollenweider's phosphorus loading diagram (Figure 19) has been developed as part of this report. These classification schemes are discussed below.

CURRENT TROPHIC STATUS CLASSIFICATION INDICES

US EPA Trophic Status Index System

The US EPA (1974d) Trophic Index System was developed as part of the National Eutrophication Survey. This system is a variation of a ranking method used by Lueschow *et al.* (1970) for 12 lakes in Wisconsin. Lueschow *et al.* used several unweighted characteristics of a water body which each reflect, in one way or another, its trophic condition. From these parameters Lueschow *et al.* derived a composite rating which was the sum of the numerical values of position for each of the parameters used in the index. The parameters used were dissolved oxygen (DO) 1 meter above bottom, organic nitrogen, total inorganic nitrogen, Secchi depth and net plankton. The water body with the lowest composite value was judged the most oligotrophic and the highest composite value lake was judged the most eutrophic.

The US EPA based their initial index system on 200+ lakes surveyed in 1972 (US EPA, 1974d). Ultimately 812 lakes will form the data base. However, rather than using the positional ranking used by Lueschow *et al.*, the US EPA adopted a percentile ranking procedure. For each of the unweighted characteristics used, the percentage of each of the 200+ lakes exceeding a given lake in that parameter (i.e., chlorophyll *a* concentration, for example) was determined. The final ranking or index value was simply the sum of the percentile ranks for each of the parameters used. The six parameters used in the US EPA Trophic Index System are summarized in Table 31. The values for the Secchi depth and minimum DO were subtracted from a fixed value (500 inches and 15 mg/l, respectively) so that all parameters would contribute a positive number to the ranking system. Using this system, a single index number was produced for each lake, so that a large number of lakes could be ranked in relative order from most oligotrophic to most eutrophic. However, this system does have several problems. This system sums the rankings for each parameter of a given water body, and thus loses information concerning specific water body characteristics. Furthermore, according to the US EPA (1974d), water bodies with very short hydraulic residence times and those with extensive littoral zones and excessive macrophyte production do not seem to fit the scheme. In the first case, the high flushing rates can cause relatively low mean nutrient concentrations in spite of high nutrient loadings. In the latter cases, the macrophytes may effectively compete with the algae for available nutrients, producing low nutrient and chlorophyll *a* levels and relatively high Secchi depths in spite of a highly eutrophic condition.

Table 31. US EPA TROPHIC STATE INDEX PARAMETERS

1. Median Total Phosphorus Concentration (mg/l).
2. Median Inorganic Nitrogen Concentration (mg/l).
3. 500 - Mean Secchi Depth (inches).
4. Mean Chlorophyll *a* (μg/l).
5. 15 - Minimum Dissolved Oxygen Concentration (mg/l).
6. Median Dissolved Phosphorus Concentration (mg/l).

Taken from US EPA (1974d).

Carlson Trophic Status Index System

Carlson's (1974) Index System is based upon Secchi depth as a means of characterizing algal biomass. As mentioned earlier, this parameter, in the absence of turbidity and colored materials in water, is a direct measure of planktonic-algal-manifested eutrophication processes in natural waters. Its range of values can easily be transformed into a convenient scale. Further, by using empirically-derived relationships between Secchi depth and both total phosphorus and chlorophyll a concentrations, Carlson has derived equations to estimate the same index value from these two parameters as well as from Secchi depth.

Carlson's Trophic Index is basically a linear transformation of Secchi depth, such that each major unit in his scale has half the value of the next lowest unit. Conversely, for total phosphorus and chlorophyll a each major unit in his scale has larger values for the next higher unit. The computational form of the equations for his trophic scheme is as follows:

$$TSI_{(SD)} = 10(6-\log_2 SD), \tag{43}$$

$$TSI_{(TP)} = 10(6-\log_2 65\ \frac{1}{TP}), \text{ and} \tag{44}$$

$$TSI_{(Chlor)} = 10(6-\log_2 7.7\ \frac{1}{Chlor^{0.68}}) \tag{45}$$

where SD = Secchi depth (m),

TP = Total phosphorus concentration (μg/l),

and Chlor = Chlorophyll a concentration (μg/l).

Calculation of the indices is facilitated by using these three equations:

$$TSI_{(SD)} = 10(6 - \frac{\ln SD}{\ln 2}), \tag{46}$$

$$TSI_{(TP)} = 10(6 - \frac{\ln \frac{65}{TP}}{\ln 2}), \text{ and} \tag{47}$$

$$TSI_{(Chlor)} = 10(6 - \frac{2.04-0.68 \ln Chlor\ a}{\ln 2} \tag{48}$$

The trophic scale and associated parameter values are presented in Table 32.

According to Carlson (1974), this index system has the advantages of easily obtained data, simplicity of form (i.e.,

Table 32. THE CARLSON TROPHIC STATE INDEX AND ITS ASSOCIATED PARAMETERS

TSI	Secchi Depth (m)	Surface Phosphorus (mg/m^3)	Surface Chlorophyll (mg/m^3)
0	64	1	0.04
10	32	2	0.12
20	16	4	0.34
30	8	8	0.94
40	4	16	2.6
50	2	32	6.4
60	1	65	20
70	0.5	140	56
80	0.25	260	154
90	0.12	519	427
100	0.062	1032	1183

Taken from Carlson (1974).

trophic condition reported as a single number), objectivity, absolute TSI values, valid relationships, retrieval of data from the index (i.e., information is not lost, as in the US EPA and Lueschow index systems) and can be intuitively grasped by the layman in much the same manner as the Richter earthquake scale.

Piwoni and Lee Trophic Status Index System

A trophic index scale has been proposed by Piwoni and Lee (1975). This index system was derived in a manner analogous to that of Lueschow *et al*. (1970), except it contains modified and additional parameters. The trophic state parameters are summarized in Table 33. The total inorganic nitrogen parameters were later dropped because the Wisconsin water bodies from which the index system was derived, and on which it was first tested, were not nitrogen-limited with respect to aquatic plan nutrient requirements.

The sum of the rankings of the water bodies, after examination of the 10 trophic index parameters, was used to classify a water body. The water body with the lowest overall ranking number was judged the most oligotrophic of the water bodies being considered. Like the US EPA (1974d) and the Lueschow *et al*. (1970) trophic index systems, the Piwoni and Lee (1975) system is a relative trophic ranking system with the water body of the highest water quality receiving the lowest trophic index number.

The Piwoni and Lee system has a significant advantage over the Lueschow *et al*. and US EPA systems in that it attempts to eliminate from the classification those parameters (characteristics) which may not properly characterize a water body's trophic state. For example, for water bodies in which the chemical nutrient determining overall algal biomass is phosphorus, (i.e., phosphorus-limited lakes) a classification system that utilizes inorganic nitrogen concentrations would incorporate extraneous information which would not be directly related to the overall water quality of the water body as it relates to excessive fertiliation.

One of the primary values of the multiparameter trophic state index system is that for a given area of the country it is possible to assess in quantitative to semi-quantitative terms the relative water quality (trophic state) of various water bodies. Lee (1974b) has utilized this approach to predict the relative water quality of a proposed impoundment, compared to other lakes and impoundments in south-central Wisconsin.

It should be noted that trophic state in a limnological sense is not directly translatable to water quality. Highly fertile water bodies in which the fertility is manifested in macrophyte growth could have a relatively low trophic state index based on the parameters normally used in the relative ranking schemes.

Table 33. PIWONI AND LEE TROPHIC STATE INDEX PARAMETERS

Parameters	Description
1. Secchi Depth	Mean of all values obtained.
2. Chlorophyll a	Average concentration in first 2 meters of water column during study period.
3. DO Depletion	Percent of lake volume with less than 0.5 mg DO/l; May to October, inclusive.
4. Winter Orthophosphate	Average in-lake concentration during winter under the ice.
5. Summer Orthophosphate	Average epilimnion concentration; May to October, inclusive.
6. Winter Total Phosphorus	Average in-lake concentrations during winter under the ice.
7. Summer Total Phosphorus	Average epilimnion concentrations; May to October, inclusive.
8. Winter Total Inorganic Nitrogen	Average in-lake concentration during winter under the ice.
9. Summer Total Inorganic Nitrogen	Average epilimnion concentration; May to October, inclusive.
10. Organic Nitrogen	Average concentration in first 2 meters of water column during study period.

Taken from Piwoni and Lee (1975).

Yet it could still have very poor water quality, if water quality is assessed in terms of impairment of beneficial uses such as swimming, boating, fishing, etc. As discussed in another section of this report, great caution should be exercised in attempting to translate the impairment in water quality associated with a given level of chlorophyll or Secchi depth from one part of the US to another. The response of the public to various degrees of algal productivity is highly subjective and regional in character (Lee, 1974b).

Rast and Lee Trophic Status Index Systems

Several approaches have been used in this study to develop a trophic index system based on the Vollenweider phosphorus loading and mean depth/hydraulic residence time relationship (Figure 19). One approach is based on the ratio of the current phosphorus loading to the permissible phosphorus loading, the latter as defined on the Vollenweider phosphorus loading diagram for a given mean depth/hydraulic residence time value (Figure 19). This approach was chosen because it reflects the amount of change in phosphorus loading necessary to attain a permissible phosphorus load for a water body with a given mean depth/hydraulic residence time relationship. Another approach was developed which relates the permissible and excessive phosphorus loads to several water quality parameters, including chlorophyll *a* and Secchi depth. These approaches are discussed below.

The first trophic index classification approach developed in this study is based on the position of the water bodies on the Vollenweider phosphorus loading diagram (Figure 19). It is reasonable to suspect that a water body which plots a large vertical distance above the permissible phosphorus loading line on Vollenweider's diagram is relatively more eutrophic than a water body which plots a smaller vertical distance above the permissible line. However, it would be inappropriate to use the linear vertical distance of a water body above the permissible phosphorus loading line because of the log-log scale of the Vollenweider diagram. The simple linear vertical distance from the permissible phosphorus loading line would not take into account that water bodies with high $\bar{z}/\tau_\omega$ values, and hence in relatively higher phosphorus loading positions on the Vollenweider diagram, would require a greater total reduction in phosphorus loads to bring them down to the permissible phosphorus loading level than would water bodies with low $\bar{z}/\tau_\omega$ values.

It should also be noted that since the permissible phosphorus loading line defines a boundary condition, it may be more appropriate to use the perpendicular displacement (i.e., shortest linear distance) of a water body from the line as a trophic ranking parameter rather than the vertical distance, particularly for water bodies with high $\bar{z}/\tau_\omega$ values. However, from the point of view of water quality management, the phosphorus loading (i.e., y-axis) is the only parameter among the Vollenweider criteria which can be controlled or managed by man. Normally, man has limited opportunity to control or manage the mean depth and hydraulic residence time of a water body. Therefore, the displacement of a water body along the y-axis (i.e., phosphorus loading) of the Vollenweider phosphorus loading diagram (Figure 19) is the parameter of concern in the Rast and Lee trophic status index.

This approach involves a determination of the magnitude of change in water quality one could expect to occur for a given change in phosphorus loading from the permissible loading level. This approach assumes that the degree of eutrophy of a water body is proportional to its phosphorus loading (i.e., phosphorus limits algal growth in the water body). While this is true for many water bodies, there are some water bodies in which phytoplankton growth is dependent on other factors such as nitrogen load. Under these conditions, the above statements would not be true over the complete range of phosphorus loads under conditions where phosphorus loads control phytoplankton growth.

This trophic index system was developed by examining whether a water body, with a certain phosphorus loading and chlorophyll a level, would experience a proportional change in water quality for a given change in phosphorus load. This can be determined by examining whether the magnitude of the phosphorus loading for a water body above the permissible phosphorus loading is matched by a proportional difference in chlorophyll a above a permissible level. As indicated in a following section of this report, the chlorophyll a concentration corresponding to the permissible phosphorus loading line on the Vollenweider diagram (Figure 19) is approximately 2 μg/l (Vollenweider, 1975a; Dillon and Rigler, 1974a; Jones and Bachmann (1976). The ratio of the current phosphorus loads to the permissible phosphorus load, as defined on the Vollenweider phosphorus diagram (Figure 19) for a given mean depth/hydraulic residence time value, was used in the trophic index. Thus, a ratio greater than one represents the excessive phosphorus loading above a certain critical phosphorus loading level for the eutrophic US OECD water bodies. Conversely, a ratio less than one represents a water body which is not receiving a "permissible" phosphorus load, relative to its mean depth/hydraulic residence time characteristics.

These ratios can be related to water quality parameters, namely chlorophyll a, Secchi depth, and hypolimnetic oxygen depletion, in order to provide trophic rankings for different water bodies. The validity of this approach stems from the fact that it has been shown in this investigation that the phosphorus load to US OECD water bodies can be highly correlated with these three parameters. These parameters are generally considered as being highly indicative of planktonic algal growth and eutrophication-related water quality.

The Rast and Lee trophic index system is similar in several respects to that of Carlson. Emphasis is placed on utilization of parameters of eutrophication (i.e. chlorophyll and Secchi depth) to which the public can generally relate. This is especially true for water clarity (i.e. Secchi depth). An important difference between the Carlson approach and this approach is that Carlson develops his trophic state index system around response parameters (i.e., chlorophyll, Secchi depth and total phosphorus). These reviewers utilized an excess nutrient loading parameter (i.e., phosphorus) as a means of classifying the relative trophic status of water bodies.

It would be of interest to develop a relationship which directly relates Vollenweider's (1975a) phosphorus loading and mean depth/hydraulic residence time diagram (Figure 19) to measurable water quality parameters, as was done with his phosphorus loading characteristics and chlorophyll a diagram (Figure 22). The development of such a model is discussed below.

This model or trophic index development centers around the loading relationship (Equation 9) which serves as the basis for the permissible phosphorus loading level in the Vollenweider diagram (Figure 19). This equation is presented below in a steady state form suitable for development of this approach:

$$L(P) = [P]_{\infty}\ \bar{z}(\rho_{\omega}+\sigma_{p}) \qquad (49)$$

where $L(P)$ = surface area total phosphorus loading (mg P/m^2/yr)

$\bar{z}$ = mean depth (m),

ρ_{ω} = hydraulic flushing rate (yr^{-1}) = $1/\tau_{\omega}$,

τ_{ω} = hydraulic residence time (yr) = water body volume (m^3)/annual inflow volume (m^3/yr),

σ_p = sedimentation coefficient for phosphorus (yr^{-1}),

and $[P]_{\infty}$ = steady state phosphorus concentration.

The same assumptions as noted for Vollenweider's model (Vollenweider, 1975a) apply to this approach. In derivation of the permissible loading line in his loading diagram, Vollenweider (1975a; 1976a) chose Sawyer's (1947) spring overturn phosphorus concentration (i.e., 10 μg/l) as the steady state phosphorus concentration in the above equation. The permissible loading line denotes the phosphorus loading, as a function of the mean depth/hydraulic residence time characteristics of a water body, which will produce a spring overturn phosphorus concentration of 10 μg/l under steady state conditions.

However, it is not mandatory that a steady state phosphorus concentration of 10 µg/l be used in Equation 49. Vollenweider chose this value "for simplicity" as a meaningful reference point around which to base boundary conditions. Other steady state phosphorus concentrations may also be used in Equation 49 to produce new phosphorus loading boundary conditions. The new boundary condition will no longer be related to Sawyer's (1947) spring overturn criteria for denoting oligotrophic versus eutrophic conditions in water bodies. Instead, the new "permissible" loading level will be the phosphorus loading which will produce the new steady state phosphorus concentration which was inserted into Equation 49.

The basis for the modification of Equation 49 in this study to relate the loading lines on the Vollenweider diagram to water quality parameters is based on earlier work by Sakamoto (1966), Dillon and Rigler (1974a) and Jones and Bachmann (1976). Dillon and Rigler (1974a; 1975), elaborating on earlier work by Sakamoto, investigated the hypothesis that a power relationship existed between chlorophyll and phosphorus in many water bodies. They correlated the summer mean chlorophyll _a_ concentration (as a measure of the algal biomass) in a water body with its spring overturn phosphorus concentration. Their data base (n=77) also included that of Sakamoto (1966) plus a number of literature values. The result was the regression equation:

$$\log_{10} [\text{chlorophyll } \underline{a}]_{\lambda}^{\text{summer}} = 1.45 \log_{10} [P]_{\lambda}^{sp} - 1.14 \quad (50)$$

where $[\text{chlorophyll } \underline{a}]_{\lambda}^{\text{summer}}$ = summer mean chlorophyll _a_ concentration (mg/m^3), and

$[P]_{\lambda}^{sp}$ = spring overturn mean total phosphorus concentration (mg/m^3).

The correlation coefficient was r=0.96, indicating a very strong relationship between these two parameters. Jones and Bachmann (1976) did a similar analysis on lakes in Iowa plus a larger number of literature values. Interestingly, Jones and Bachmann regressed the summer mean chlorophyll concentration on the summer mean total phospohrus, rather than the spring overturn total phosphorus. However, they obtained an almost identical regression equation and correlation coefficient:

$$\log_{10} [\text{chlorophyll } \underline{a}]_{\lambda}^{\text{summer}} = 1.46 \log_{10} [P]_{\lambda} - 1.09 \quad (51)$$

$$r = 0.95$$

This indicates a water body's total phosphorus concentration appears to remain relatively constant over the annual cycle. Such an occurrence was demonstrated by Lee _et al_. (1976) in studies on Lake Mendota.

One may incorporate the work of Dillon and Rigler (1974a; 1975) and Jones and Bachmann (1976) into Vollenweider's equation for the permissible and/or excessive phosphorus loading levels to produce boundary conditions manifested in a water quality parameter, namely phosphorus concentration. Equation 50 above can be rearranged as:

$$\log_{10} [P]_{\lambda}^{sp} = \frac{\log_{10} [\text{chlorophyll } \underline{a}]^{\text{summer}} + 1.14}{1.45}, \qquad (52)$$

Equation 49 can be arranged in the same manner as:

$$\log_{10} [P]_{\lambda}^{\text{summer}} = \frac{\log_{10} [\text{chlorophyll } \underline{a}]^{\text{summer}} + 1.09}{1.46}. \qquad (53)$$

One can solve these equations for the phosphorus concentrations which will produce a given summer chlorophyll a concentration. A useful point about the above equations is that one can solve them to obtain the relationship between as many total phosphorus and chlorophyll a concentrations as desired. The result will be a sequence of different total phosphorus and chlorophyll a data sets.

The final step in the development of this approach is to use either Equation 52 or 53, or the mean value of both equations, and Equation 49 to translate Vollenweider's permissible and excessive loading lines (Figure 19) into expected chlorophyll a concentrations. This can be done by the use of Equations 52 and/or 53 to determine the total phosphorus concentration required to produce a given summer epilimnetic chlorophyll a concentration. The resultant phosphorus concentration can be inserted into Equation 49, which can then be solved for the particular phosphorus loading necessary to produce the inserted phosphorus concentration. In addition, the phosphorus concentration has also been related to a chlorophyll a concentration (Equations 50 and/or 51). Thus, the solution of Equation 49 for a given steady state phosphorus concentration also directly relates the phosphorus loading to a given chlorophyll a concentration. One can then also relate these boundary lines to Secchi depth and hypolimnetic oxygen depletion with the use of Equations 38 and 41, respectively. With the use of these equations Vollenweider's phosphorus loading diagram (Figure 19) can be transformed so as to relate phosphorus loads to summer chlorophyll, Secchi depth and hypolimnetic oxygen depletion conditions in a water body. The permissible and excessive phosphorus loading lines correspond, based on spring overturn phosphorus concentrations of 10 μg/l and 20 μg/l, respectively (Sawyer, 1947; Sakamoto, 1966; Dillon and Rigler, 1975), to summer mean epilimnetic chlorophyll a concentrations of 2 μg/l and 6 μg/l, to mean Secchi depths of 4.6 m and 2.7 m and hypolimnetic oxygen depletion rates of 0.3 mg O_2/m^2/day and 0.6 mg O^2/m^2/day, respectively. The results of the above approach are presented in a following section.

TROPHIC STATUS INDICES AS APPLIED TO THE US OECD WATER BODIES

US EPA Trophic Status Index System

The US EPA (1974d) Trophic State Index parameters were listed in Table 31. Because the minimum dissolved oxygen concentration was not available for most of the US OECD water bodies, this parameter (i.e., 15 minus the dissolved oxygen concentration) was not included in the final ranking number. While this means that the final ranking of the US OECD water bodies is based only on five of the six US EPA Trophic State Index parameters, it should still give a reasonably accurate relative trophic state ranking of the US OECD water bodies. For the purpose of this discussion, the US EPA approach is described as "modified" (ommission of DO value) from the classification scheme. In general, the data used in the US EPA Trophic State Index, as well as that of Carlson, Piwoni and Lee, and Rast and Lee, was taken from the US OECD Summary Sheets (Appendix II) in this report. However, Rast and Lee also made use of Vollenweider's phosphorus loading diagram (Figure 19).

The relative ranks of the US OECD water bodies based on the five US EPA trophic state index parameters used in this classification effort are presented in Table 34. In this system, the water bodies with the lowest trophic status index number are relatively the most productive, while the least productive lake in the series will have the highest trophic status index number. The US EPA (1974d) used the percent of the lakes in their study which exceeded a parameter value for each lake to produce the relative ranking for each lake for a given trophic state index parameter. The same method was used by these reviewers, but the actual number of lakes, rather than the percent exceeding a parameter value for a particular lake, was used in the ranking. The relative ranking position of the water bodies is identical in both cases. Water bodies with identical parameter values (ties) were given the same ranking number. It should be noted that all parameter values were not available for all US OECD water bodies.

The trophic status rankings of the US OECD water bodies, using the modified US EPA criteria, are presented in Table 35. Since no trophic condition has been associated with a particular Trophic Status Index Number(s), the trophic ranking is by necessity only a relative ranking. In general, the relative trophic ranking of the US OECD water bodies is as expected based on the relative general characteristics of the water bodies. There are, however, several anomalies in the ranking, based on the trophic conditions reported by the US OECD investigators. Particularly, Lakes Harriet, Washington-1957, Calhoun, and Shagawa appear to be higher in the ranking (i.e., more toward the oligotrophic end of the scale) than expected. Conversely, it would be said that Lakes Cayuga and Weir are lower in the ranking than would be expected, relative to the reported trophic conditions for the other water bodies in the ranking. These apparent anomalies will be addressed in more detail in a following section.

Table 34. RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATE INDEX SYSTEM

	Relative Ranking Under Indicated Parameter:					Trophic Status
Water Body	Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[e]	500-Secchi Depth (inches)	Chlorophyll a (µg/l)	Dissolved Phosphorus (mg/l)	Index Number (Sum of Rankings)
Blackhawk (E)[a]	11[b]	6	43	19[i]	11	90
Brownie (E)	-	39[c,f]	14	35[c]	25	-
Calhoun (E)	8[c]	33[c,f]	24	34[c]	34	133
Camelot-Sherwood (E)	29[b]	4	23	29[i]	27	112
Canadarago (E)						
1968	26	18	20	22	19	105
1969	28	15	20	28	19	110
Cayuga (M)						
1972	36	20	33	34	38	161
1973	36	12	33	38	36	155
Cedar (E)	24[c]	39[c,f]	20	15[c]	34	132
Cox Hollow (E)	16[b]	8	14	8[i]	10	56
Dogfish (O)						
1971	44	17	36	34	-	-
1972	44	-	35	39	-	-
Dutch Hollow (E)	3[b]	16	2	4[i]	20	45
George (O-M)	46	40[f]	45	-	40	-
Harriet (E)	20[c]	39[c,f]	34	40[c]	34	167
Isles (E)	7[c]	39[c,f]	3	2[c]	25	76
Kerr Reservoir (E-M)						
Roanoke Arm	32	22	14	20	23	111
Nutbush Arm	32	25	6	12	19	94

Table 34 (continued). RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATE INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[e]	500-Secchi Depth (inches)	Chlorophyll a (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)
Lamb (O)						
1971	40	12	20	34	-	-
1972	42	-	27	41	-	-
Meander (O)						
1971	42	13	40	38	-	-
1972	45	-	39	43	-	-
Mendota (E)	4	7	39	26	3	79
Michigan						
Openwaters - 1971 (O)	39	30	33	43	42	187
Nearshore Waters - 1971(M)	38	27	-	38	41	-
Lower Minnetonka						
1969 (E)	23	-	14	13	-	-
1973 (E→M)	26	-	20	25	38	-
Potomac Estuary (U-E)						
Upper Reach	0[d]	1[d]	0[d]	0[d]	0[d]	1
Middle Reach	1[d]	24[d]	2[d]	1[d]	3[d]	31
Lower Reach	27[d]	34[d]	14[d]	18[d]	14[d]	107
Redstone (E)	17[b]	12	16	21[i]	27	93
Sallie (E)	2	15	-	-	1	-
Sammamish (M)	32	29[g]	41	-	29	-
Shagawa (E)	23	31	33	18	14	119
Stewart (E)	23[b]	0	8	23[i]	36	90

Table 34 (continued). RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATE INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[e]	500-Secchi Depth (inches)	Chlorophyll a (µg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)
Tahoe (U-O)	47	41	47	45[j]	34	214
East Twin						
1972 (E)	16	9	16	9	10	60
1973 (E)	16	2	33	11	7	69
1974 (E)	16	-	22	7	19	-
West Twin						
1972 (E)	5	5	27	3	5	45
1973 (E)	7	4	37	10	5	63
1974 (E)	9	-	33	7	7	-
Twin Valley (E)	19[b]	20	14	16[i]	22	91
Virginia (E)	11[b]	27	6	5[i]	22	71
Waldo (U-O)	48[h]	42[h]	46[h]	44[h]	34[h]	214
Washington						
1957 (E)	33	32	27	25	40	157
1964 (E)	18	24	6	15	10	73
1971 (M)	37	29	42	34	29	171
1974 (M)	-	-	44	-	-	-
Weir (M)	16	35	22	27	14	114
Wingra (E)	36	21[f]	7	-	19	-

Table 34 (continued). RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATE INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[e]	500-Secchi Depth (inches)	Chlorophyll $\underline{a}$ (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)

EXPLANATION:

[a]Investigator-indicated trophic condition

E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

[b]Based on mean of summer and winter concentrations.

[c]Based on mean summer surface values.

[d]Based on mean summer values.

[e]Based on $NH_4^+ + NO_3^- + NO_2^-$ (as N) unless otherwise indicated.

[f]Based on $NH_4^+ + NO_3^-$ (as N) values.

[g]Based on $NO_3^- + NO_2^-$ (as N) values.

[h]Based on August values from 1970 to 1974.

[i]Based on samples from upper two meters of water column.

[j]Based on euphotic zone measurements.

Dash (-) indicates data not available.

Table 35. RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATUS INDEX SYSTEM.

	Water Body	Investigator-Indicated Trophic Status	Trophic Status Index Number
tie	Tahoe	ultra - oligotrophic	214
tie	Waldo	ultra - oligotrophic	214
	Michigan Open Waters - 1971	oligotrophic	187
	Washington - 1971	mesotrophic	171
	Harriet	eutrophic	167
	Cayuga - 1972	mesotrophic	161
	Washington - 1957	eutrophic	157
	Cayuga - 1973	mesotrophic	155
	Calhoun	eutrophic	133
	Shagawa	eutrophic	119
	Weir	mesotrophic	114
	Camelot - Sherwood	eutrophic	112
	Kerr - Roanoke Arm	eutrophic - mesotrophic	111
	Canadarago - 1969	eutrophic	110
	Potomac - Lower Reach	eutrophic	107
	Canadarago - 1968	eutrophic	105
	Kerr - Nutbush Arm	eutrophic - mesotrophic	94
	Redstone	eutrophic	93
	Twin Valley	eutrophic	91
tie	Blackhawk	eutrophic	90
tie	Stewart	eutrophic	90
	Mendota	eutrophic	79
	Isles	eutrophic	76
	Washington - 1964	eutrophic	73
	Virginia	eutrophic	71
	East Twin - 1973	eutrophic	69
	West twin - 1973	eutrophic	63

Table 35 (continued). RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING MODIFIED US EPA TROPHIC STATUS INDEX SYSTEM

	Water Body	Investigator-Indicated Trophic Status	Trophic Status Index Number
	East Twin - 1972	eutrophic	60
	Cox Hollow	eutrophic	45
tie	Dutch Hollow	eutrophic	45
tie	West Twin - 1972	eutrophic	45
	Potomac - Middle Reach	ultra-eutrophic	31
	Potomac - Upper Reach	ultra-eutrophic	1

Carlson Trophic Status Index System

The parameters in Carlson's (1974) Trophic Status Index were listed in Table 32. An absolute TSI value can be assigned to a water body on the basis of either its phosphorus or chlorophyll concentration and/or its Secchi depth. However, the trophic rankings are still relative, as with the US EPA Trophic Status Index System, since no TSI value or range was assigned to a given trophic condition in Carlson's system. If it were necessary to assign a TSI value to a given trophic condition, general limnological knowledge would suggest that a reasonable boundary value between eutrophic and oligotrophic might be a TSI value of 40. This TSI value would indicate a Secchi depth of 4 meters, a chlorophyll concentration of 2.6 µg/l and a phosphorus concentration of about 16 µg/l. This value as a boundary condition is based solely on the experience of these reviewers.

The relative rankings of the US OECD water bodies, based on their phosphorus and chlorophyll concentrations, and Secchi depths, are presented in Table 36. Inspection of Table 36 indicates that the relative positions of the US OECD water bodies vary widely, depending on the particular Carlson TSI parameter examined.

In order to demonstrate the relationship between the three Carlson TSI parameters, the US OECD water bodies were ranked by these parameters on the basis of increasing productivity or eutrophy. In this ranking the order of water bodies is from the oligotrophic end of the trophic scale to the eutrophic end, with the relatively most eutrophic water body listed first. The results are presented in Table 37 (the investigator-indicated trophic states were indicated in Table 36).

A general inspection of Table 37 shows that while the ultra-oligotrophic and ultra-eutrophic water bodies are listed at the appropriate ends of the ranking scales, there are a number of differences in the relative positions of US OECD water bodies using the three different Carlson TSI parameters. For example, the reported Secchi depths for Lakes Blackhawk, Mendota, and Harriet place them higher (i.e., toward the oligotrophic end of the scale) in the relative ranking than several other water bodies generally considered less productive (i.e., Washington - 1971, Dogfish and Cayuga, respectively). The chlorophyll concentrations for Lakes Harriet, Brownie and Calhoun also place them higher in the relative ranking than less productive Lakes Cayuga, Dogfish and Lamb.

Carlson (1974), using data for Lake Washington, has demonstrated that the data for this lake and TSI values follow the same trends and that they produce the same relative values when transformed to the trophic scale. He has also indicated that the TSI values (and relative rankings) are not always identical. Such an anomaly can be used as an internal check on the assumptions being

Table 36. RANKING OF US OECD WATER BODIES USING CARLSON TROPHIC STATUS INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: TSI(SD)	TSI(TP)	TSI(Chlorophyll)
Blackhawk (E)[a]	5	35[b]	26[f]
Brownie (E)	33	-	12
Calhoun (E)	24	37[c]	13[c]
Camelot-Sherwood (E)	25	19[b]	18[f]
Canadarago (E)			
1968	-	22	25
1969	28	20	19
Cayuga (M)			
1972	15	13	13
1973	15	13	8
Cedar (E)	28	24[c]	29[c]
Cox Hollow (E)	33	31[b]	37[f]
Dogfish (O)			
1971	12	5	13
1972	13	5	7
Dutch Hollow (E)	45	43[b]	41[f]
George (O-M)	3	3	-
Harriet (E)	14	28[c]	6[c]
Isles (E)	44	38	43[c]
Kerr Reservoir (E-M)			
Roanoke Arm	33	16	26
Nutbush Arm	41	16	33
Lamb (O)			
1971	28	9	13
1972	21	7	5
Meander (O)			
1971	8	7	8
1972	9	4	3
Mendota (E)	9	42	21

Table 36(continued). RANKING OF US OECD WATER BODIES USING CARLSON TROPHIC INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: TSI(SD)	TSI(TP)	TSI(Chlorophyll)
Michigan (O-M)			
Nearshore Waters-1971	15	11	8
Open Waters-1971	-	9	3
Lower Minnetonka			
1969 (E)	33	25	32
1973 (E→M)	28	22	22
Potomac Estuary (U-E)			
Upper Reach	47[d]	46[d]	45[d]
Middle Reach	45[d]	45[d]	44[d]
Lower Reach	33[d]	21[d]	27[d]
Redstone (E)	31	30[b]	25[f]
Sallie (E)	-	44	-
Sammamish (M)	7	16	8
Shagawa (E)	15	25	27
Stewart (E)	33	25[b]	24[f]
Tahoe (U-O)	1	1	1[g]
East Twin			
1972 (E)	31	31	36
1973 (E)	15	31	34
1974 (E)	26	31	38
West Twin			
1972 (E)	21	41	42
1973 (E)	11	38	35
1974 (E)	15	38	38
Twin Valley (E)	33	29[b]	29[f]
Virginia (E)	41	35[b]	40[f]
Waldo (U-O)	2[e]	2[e]	2[e,g]

Table 36(continued). RANKING OF US OECD WATER BODIES USING CARLSON TROPHIC INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: TSI(SD)	TSI(TP)	TSI(Chlorophyll)
Washington			
1957 (E)	21	15	22
1964 (E)	41	29	29
1971 (M)	6	12	13
1974 (M)	4	-	-
Weir (M)	26	31	20
Wingra (E)	40	13	-

EXPLANATION:

[a]Investigator-indicated trophic status:

E = eutrophic
M = mesotrophic
O = oligotrophic
U = ultra

[b]Based on mean of summer and winter concentrations.

[c]Based on mean summer surface values.

[d]Based on mean summer values.

[e]Based on August values from 1970 to 1974.

[f]Based on samples from upper two meters of water column.

[g]Based on euphotic zone measurements.

Dash (-) indicates data not available.

Table 37. RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING CARLSON TROPHIC STATUS INDEX SYSTEM

TSI(SD)	TSI(TP)	TSI(Chlorophyll)
Tahoe	Tahoe	Tahoe
Waldo	Waldo	Waldo
George	George	tie: Meander-1972 Michigan Open Waters-1971
Washington-1974	Meander-1972	Lamb-1972
Blackhawk	tie: Dogfish-1971 Dogfish-1972	Harriet
Washington-1971	tie: Lamb-1972 Meander-1972	Dogfish-1972
Sammamish	Lamb-1971	tie: Cayuga-1973 Meander-1971 Michigan, Nearshore Waters-1971 Sammamish
Meander-1971	Michigan Open Waters-1971	Brownie
tie: Meander-1972 Mendota	Michigan, Nearshore Waters-1971	tie: Calhoun Cayuga-1972 Dogfish-1971 Lamb-1971 Washington-1971
West Twin-1973	Washington-1971	Camelot-Sherwood
Dogfish-1971	tie: Cayuga-1972 Cayuga-1973	Canadarago-1969
Dogfish-1972	Washington-1957	Weir
Harriet	tie: Kerr-Roanoke Arm Kerr-Nutbush Arm Sammamish	Mendota
tie: Cayuga-1972 Cayuga-1973 Michigan Open Waters - 1971 Shagawa East Twin-1973 West Twin-1974	Camelot-Sherwood	tie: Lower Minnetonka-1973 Washington-1957
tie: Lamb-1972 West Twin-1972 Washington-1957	Canadarago-1969	Stewart
Calhoun	Potomac-Lower Reach	Canadarago-1968
Camelot-Sherwood	tie: Canadarago-1968 Lower Minnetonka-1973	
tie: East Twin-1974 Weir	Cedar	

Table 37(continued) RELATIVE TROPHIC STATUS RANKING OF THE US OECD WATER BODIES USING CARLSON TROPHIC STATUS INDEX SYSTEM

TSI(SD)	TSI(TP)	TSI(Chlorophyll)
tie: Canadarago-1969 Cedar Lamb-1971 Lower Minnetonka-1973	tie: Lower Minnetonka-1969 Shagawa Stewart	Redstone
tie: Redstone East Twin-1972	Harriet	Kerr-Roanoke Arm
Brownie	Twin Valley	Blackhawk
tie: Cox Hollow Kerr-Roanoke Arm Lower Minnetonka-1969 Potomac-Lower Reach Twin Valley	Washington-1964	tie: Potomac-Lower Reach Shagawa
Stewart	Redstone	tie: Cedar Twin Valley Washington-1964
Wingra	tie: Cox Hollow East Twin-1972 East Twin-1973 East Twin-1974	Lower Minnetonka-1969
tie: Kerr-Nutbush Arm Virginia Washington-1964	tie: Blackhawk Virginia	Kerr-Nutbush Arm
Isles	Calhoun	East Twin-1973
tie: Dutch Hollow Potomac-Middle Reach	tie: Isles West Twin-1973 West Twin-1974	West Twin-1973
Potomac-Upper Reach	West Twin-1972	East Twin-1972
	Mendota	Cox Hollow
	Dutch Hollow	tie: East Twin-1974 West Twin-1974
	Sallie	Virginia
	Potomac-Middle Reach	Dutch Hollow
	Potomac-Upper Reach	West Twin-1972
		Isles
		Potomac-Middle Reach
		Potomac-Upper Reach

made about a water body's utilization of phosphorus for planktonic algal growth. To cite an example (Carlson, 1974), if a water body has a higher TSI(TP) than its TSI(SD) and TSI (Chlorophyll), and the latter two values are similar, then it may indicate that the water body is not phosphorus-limited.

Piwoni and Lee Trophic Status Index System

The Trophic State Index parameters of Piwoni and Lee were presented in Table 33. As with the US EPA (1974d) Trophic State Index System, all parameter values were not available for all US OECD water bodies. As before, if a water body did not have values for all the Piwoni and Lee Trophic State Index parameters, it was not included in the final ranking. Further, the parameters used by these reviewers in ranking the US OECD water bodies using the Piwoni and Lee system were altered so that available data could be used. Phosphorus and nitrogen values were not reported on a seasonal basis in most cases. Also, the DO depletion was unavailable for most US OECD water bodies. The result was that the Secchi depths, total and dissolved phosphorus, and inorganic nitrogen concentration, and chlorophyll a concentration of the US OECD water bodies were used to rank them in the Piwoni and Lee Trophic Status Index System. The rankings of the US OECD water bodies using the modified Piwoni and Lee parameters, are presented in Table 38.

The relative ranks of the US OECD water bodies, based on the five modified Piwoni and Lee Trophic State Index parameters, are presented in Table 39. In this table, the more oligotrophic water bodies are listed first. As with the other relative trophic rankings, there is general agreement between the US OECD water body's relative trophic rankings and the trophic conditions indicated by their respective investigators. Lake Harriet occupies a higher relative ranking than less productive Lakes Washington - 1974 and Cayuga, while Lake Weir occupies a lower relative ranking than more productive Lakes Cedar, Shagawa and Calhoun. Lake Shagawa, based on limnological characteristics, also occupies a higher ranking than less productive water bodies.

Rast and Lee Trophic Status Index System

For this discussion, these authors chose several of the same trophic state indicators used in Carlson's (1974) Trophic Status Index System; namely Secchi depth and chlorophyll a. The ranking of the US OECD water bodies, based on their current phosphorus loading/permissible phosphorus loading and current chlorophyll/permissible chlorophyll quotients, Secchi depth and chlorophyll a concentrations as ranking parameters, is presented in Table 40. The relative rankings of the US OECD water bodies, based on the above-mentioned parameters, are listed in Table 41.

Table 38. RANKING OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[h]	Secchi Depth (m)	Chlorophyll a (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)
Blackhawk (E)[a]	39[g]	36[g]	5	27[d]	32[g]	139
Brownie (E)	-	4[e,i]	34	11[e]	18[e]	-
Calhoun (E)	42[e]	11[e,i]	24	12[e]	9[e]	98
Camelot-Sherwood (E)	20[g]	38[g]	25	17[d]	16[g]	116
Canadarago (E)						
1968	23	25	28	24	24	124
1969	21	28	-	18	24	-
Cayuga (M)						
1972	13	23	15	12	5	68
1973	13	31	15	8	7	74
Cedar (E)	25[e]	4[e,i]	28	31[e]	9[e]	97
Cox Hollow (E)	33[g]	34[g]	34	38[d]	33[g]	172
Dogfish (O)						
1971	5	26	12	12	-	-
1972	5	-	13	7	-	-
Dutch Hollow (E)	47[g]	27[g]	46	42[d]	23[g]	185
George (O-M)	3	3[i]	3	-	3	-
Harriet (E)	29[e]	4[e,i]	14	6[e]	9[e]	62
Isles (E)	43[e]	4[e,i]	45	44[e]	18[e]	154
Kerr Reservoir (E-M)						
Roanoke Arm	17	21	34	26	20	118
Nutbush Arm	17	18	42	34	24	135

Table 38 (continued). RANKING OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l) [h]	Secchi Depth (m)	Chlorophyll a (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)
Lamb (O)						
1971	9	31	28	12	-	-
1972	7	-	21	5	-	-
Meander (O)						
1971	7	30	8	8	-	-
1972	4	-	9	3	-	-
Mendota (E)	46	35	9	20	46	156
Michigan						
Nearshore Waters (M) -1971	11	16	-	8	2	-
Open Waters (O) -1971	10	13	15	3	1	42
Lower Minnetonka						
1969 (E)	26	-	34	33	-	-
1973 (E-M)	23	-	28	21	5	-
Potomac Estuary (U-E)						
Upper Reach	50[b]	42[b]	48[b]	46[b]	43[b]	229
Middle Reach	49[b]	19[b]	46[b]	45[b]	40[b]	199
Lower Reach	22[b]	10[b]	34[b]	28[b]	30[b]	124
Redstone (E)	32[g]	32[g]	32	25[d]	16[g]	137
Sallie (E)	48	28	-	-	42	-
Sammamish (M)	17	14[j]	7	8	14	60
Shagawa (E)	26	13	15	28	29	111
Stewart (E)	26[g]	41[g]	40	23[d]	7[g]	137

Table 38(continued). RANKING OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)[h]	Secchi Depth (m)	Chlorophyll a (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)
Tahoe (U-O)	1	2	1	1	9	14
East Twin						
1972 (E)	33	33	32	37	33	168
1973 (E)	33	40	15	35	36	159
1974 (E)	33	-	26	39	24	-
West Twin						
1972 (E)	45	37	21	43	38	184
1973 (E)	43	38	11	36	38	166
1974 (E)	41	-	15	39	36	-
Twin Valley (E)	30[g]	23[g]	34	30[d]	21[6]	138
Virginia (E)	39[g]	16[g]	42	41[d]	21[g]	159
Waldo (U-O)	2[c]	1[c]	2[c]	2[c,f]	9[c]	16
Washington						
1957 (E)	16	12	21	21	3	73
1964 (E)	31	19	42	31	33	156
1971 (M)	12	14	6	12	14	58
1974 (M)	13	22[i]	41	-	24	-

EXPLANATION:

[a]Investigator-indicated trophic status
- E = eutrophic
- M = mesotrophic
- O = oligotrophic
- U = ultra

Table 38(continued). RANKING OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

Water Body	Relative Ranking Under Indicated Parameter: Total Phosphorus (mg/l)	Inorganic Nitrogen (mg/l)	Secchi Depth (m)	Chlorophyll *a* (μg/l)	Dissolved Phosphorus (mg/l)	Trophic Status Index Number (Sum of Rankings)

[b]Based on mean summer values.

[c]Based on August values from 1970 to 1974.

[d]Based on samples taken from upper two meters of water column.

[e]Based on mean summer surface values.

[f]Based on eutrophic zone measurements.

[g]Based on mean of summer and winter concentrations.

[h]Based on NH_4^++NO_3^-+NO_2^- (as N) unless otherwise indicated.

[i]Based on NH_4^++NO_3^- (as N) values.

[j]Based on NO_3^-+NO_2^- (as N) values.

Table 39. RELATIVE TROPHIC STATUS RANKINGS OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

	Water Body	Investigator-Indicated Trophic Status	Trophic Status Index Number
	Tahoe	ultra-oligotrophic	14
	Waldo	ultra-oligotrophic	16
	Michigan Open Waters - 1971	oligotrophic	42
	Washington - 1971	mesotrophic	58
	Sammamish	mesotrophic	60
	Harriet	eutrophic	62
	Cayuga - 1972	mesotrophic	68
	Cayuga - 1973	mesotrophic	74
	Washington - 1974	mesotrophic	82
	Cedar	eutrophic	97
	Calhoun	eutrophic	98
	Shagawa	eutrophic	111
	Camelot-Sherwood	eutrophic	116
	Weir	mesotrophic	117
	Kerr-Roanoke Arm	eutrophic-mesotrophic	118
tie	Canadarago - 1968	eutrophic	124
	Potomac - Lower Reach	eutrophic	124
	Kerr - Nutbush Arm	eutrophic-mesotrophic	135
tie	Redstone	eutrophic	137
	Stewart	eutrophic	137
	Twin Valley	eutrophic	138
	Virginia	eutrophic	144
	Mendota	eutrophic	150
	Isles	eutrophic	154
	Washington - 1964	eutrophic	156
	East Twin - 1973	eutrophic	159
	West Twin - 1973	eutrophic	166

Table 39 (Continued). RELATIVE TROPHIC STATUS RANKINGS OF US OECD WATER BODIES USING PIWONI AND LEE MODIFIED TROPHIC STATUS INDEX SYSTEM

Water Body	Investigator-Indicated Trophic Status	Trophic Status Index Number
East Twin - 1972	eutrophic	168
West Twin - 1972	eutrophic	184
Dutch Hollow	eutrophic	185
Potomac - Middle Reach	ultra-eutrophic	199
Potomac - Upper Reach	ultra-eutrophic	229

Table 40. RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL AND EXCESS PHOSPHORUS LOADING AS RANKING PARAMETERS

Water Body	Relative Ranking Under Indicated Parameters			
	Mean Secchi Depth (m)	Mean Chlorophyll a (µg/l)	(Current Chlorophyll a)/(Permissible Chlorophyll a)	(Current Phosphorus Loading)/(Permissible Phosphorus Loading)
Blackhawk (E)[a]	5	26[c]	27	41
Brownie (E)	33	12[d]	14	36
Calhoun (E)	24	13[d]	15	32
Camelot-Sherwood (E)	25	18[c]	18	33
Canadarago (E)				
1968	-	25	19	-
1969	28	19	22	20
Cayuga (M)				
1972	15	13	20	26
1973	15	8	13	-
Cedar (E)	28	29[d]	31	17
Cox Hollow (E)	33	37[c]	38	42
Dogfish (O)				
1971	12	13	7	-
1972	13	7	4	2
Dutch Hollow (E)	45	41[c]	42	34
George (O-M)	3	-	-	6
Harriet (E)	14	6[d]	7	28
Isles (E)	44	43[d]	44	44
Kerr Reservoir (E-M)				
Roanoke Arm	33	26	26	35
Nutbush Arm	41	33	29	30
Lamb (O)				
1971	28	13	11	-
1972	21	5	7	3

Table 40 (continued). RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL AND EXCESS PHOSPHORUS LOADING AS RANKING PARAMETERS

Water Body	Relative Ranking Under Indicated Parameters			
	Mean Secchi Depth (m)	Mean Chlorophyll a (μg/l)	(Current Chlorophyll a)/(Permissible Chlorophyll a)	(Current Phosphorus Loading)/(Permissible Phosphorus Loading)
Meander (O)				
1971	8	8	7	-
1972	9	3	3	3
Mendota (E)	9	21	31	37
Michigan (O) -open waters (τ_{ω} = 30 yrs)				
1971	-	3	5	11
1974	-	-	-	8
Michigan (O) -open waters (τ_{ω} = 100 yrs)				
1971	-	3	5	12
1974	-	-	-	9
Lower Minnetonka				
1969 (E)	33	32	33	25
1973 (E→M)	28	22	21	10
Potomac Estuary (U-E)				
Upper Reach	47[b]	45[b]	46	48
Middle Reach	45[b]	44[b]	45	47
Lower Reach	33[b]	27[b]	28	31
Redstone (E)	31	25[c]	25	40
Sallie (E)	-	-		46
Sammamish (M)	7	8	12	20
Shagawa (E)	15	27	35	23

Table 40 (continued). RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL AND EXCESS PHOSPHORUS LOADING AS RANKING PARAMETERS

Water Body	Relative Ranking Under Indicated Parameters			
	Mean Secchi Depth (m)	Mean Chlorophyll a (μg/l)	(Current Chlorophyll a)/(Permissible Chlorophyll a)	(Current Phosphorus Loading)/(Permissible Phosphorus Loading)
Stewart (E)	33	24[c]	24	45
Tahoe (U-O)	1	1[f]	1	5
East Twin				
1972 (E)	31	36	37	24
1973 (E)	15	34	34	19
1974 (E)	26	38	39	20
West Twin				
1972 (E)	21	42	43	18
1973 (E)	11	35	36	16
1974 (E)	15	38	39	15
Twin Valley (E)	33	29[c]	30	39
Virginia (E)	41	40[c]	41	43
Waldo (U-O)	2[e]	2[e,f]	1	1
Washington				
1957 (E)	21	22	23	27
1964 (E)	41	29	31	38
1971 (M)	6	13	16	13
1974 (M)	4	-	-	14
Weir (M)	26	20	17	6
Wingra (E)	40	-	-	29

Explanation:

[a]Investigator-indicated trophic state:

E = eutrophic
M - mesotrophic
O = oligotrophic
U = ultra

Table 40 (continued). RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL AND EXCESS PHOSPHORUS LOADING AS RANKING PARAMETERS

EXPLANATION (continued)

[b]Based on mean summer values.

[c]Based on samples taken from upper two meters of water column.

[d]Based on summer surface values.

[e]Based on August values from 1970 to 1974.

[f]Based on euphotic zone measurements.

[g]Based on mean of summer and winter concentrations.

Dash (-) indicates data not available.

Table 41. RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL a AND EXCESS PHOSPHORUS LOAD

Mean Secchi Depth (m)	Mean Chlorophyll a ($\mu g/l$)	(Current Chlorophyll a)/(Permissible Chlorophyll a)	(Current Phosphorus Load)/(Permissible Phosphorus Load)
Tahoe	Tahoe	Tahoe	Waldo
Waldo	Waldo	Waldo	Dogfish - 1972
George	Meander - 1972	Meander - 1972	Lamb - 1972
Washington - 1974	Michigan - Open Waters - 1971	Dogfish - 1972	Meander - 1972
Blackhawk	Lamb - 1972	Michigan Open Waters - 1971 (τ_ω = 30 & 100 yrs)	Tahoe
Washington - 1971	Harriet	Harriet	George
Sammamish	Dogfish - 1972	Lamb - 1972	Weir
Meander - 1971	Cayuga - 1973	Meander - 1971	Michigan Open Waters - 1974 (τ_ω = 30 yrs)
Meander - 1972	Meander - 1971	Dogfish - 1971	Michigan Open Waters - 1974 (τ_ω = 100 yrs)
Mendota	Michigan Nearshore Waters - 1971	Lamb - 1971	Minnetonka - 1973
West Twin - 1973	Brownie	Dogfish - 1971	Michigan Open Waters - 1971 (τ_ω = 30 yrs)
Dogfish - 1971	Calhoun	Sammamish	Michigan Open Waters - 1971 (τ_ω = 100 yrs)
Dogfish - 1972	Cayuga - 1972	Cayuga - 1973	Washington - 1971
Harriet	Dogfish - 1971	Brownie	
	Lamb - 1971	Calhoun	
	Washington - 1971	Washington - 1971	
		Weir	
		Camelot-Sherwood	

Table 41 (continued). RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL *a*, EXCESS CHLOROPHYLL *a* AND EXCESS PHOSPHORUS LOAD

Mean Secchi Depth (m)	Mean Chlorophyll *a* (μg/l)	(Current Chlorophyll *a*)/(Permissible Chlorophyll *a*)	(Current Phosphorus Load)/(Permissible Phosphorus Load)
Cayuga - 1972	Camelot-Sherwood	Canadarago - 1968	Washington - 1974
Cayuga - 1973	Canadarago - 1969	Cayuga - 1972	West Twin - 1974
Michigan Open Waters-1971	Weir	Minnetonka - 1973	West Twin - 1973
Shagawa	Mendota	Canadarago - 1969	Cedar
East Twin - 1973	Lower Minnetonka 1973	Washington - 1957	West Twin - 1972
West Twin - 1974	Washington - 1957	Stewart	East Twin - 1973
Lamb - 1972	Stewart	Redstone	Canadarago
West Twin - 1972	Canadarago - 1968	Kerr Reservoir -Roanoke Arm	Sammamish
Washington - 1957	Redstone	Blackhawk	East Twin - 1974
Calhoun	Kerr-Roanoke Arm	Kerr Reservoir -Nutbush Arm	Shagawa
Camelot-Sherwood	Blackhawk	Potomac Estuary -Lower Reach	East Twin - 1972
East Twin - 1974	Potomac-Lower Reach	Kerr Reservoir -Nutbush Arm	Minnetonka - 1969
Weir	Shagawa	Twin Valley	Cayuga
Canadarago - 1969	Twin Valley	Cedar	Washington - 1957
	Cedar	Mendota	Harriet
	Washington-1964	Minnetonka - 1969	Wingra
	Lower Minnetonka 1969	East Twin - 1973	Kerr Reservoir - Nutbush Arm
	Kerr-Roanoke Arm	Shagawa	Potomac Estuary - Lower Reach
			Calhoun

Table 41 (continued). RELATIVE TROPHIC STATUS RANKING OF US OECD WATER BODIES USING SECCHI DEPTH, CHLOROPHYLL a, EXCESS CHLOROPHYLL a AND EXCESS PHOSPHORUS LOAD

Mean Secchi Depth (m)	Mean Chlorophyll a ($\mu g/l$)	(Current Chlorophyll a)/ (Permissible Chlorophyll a)	(Current Phosphorus Load)/ (Permissible Phosphorus Load)
Cedar	East Twin - 1973	West Twin - 1973	Camelot-Sherwood
Lamb - 1971		East Twin - 1972	Dutch Hollow
Lower Minne-tonka - 1973		Cox Hollow	Kerr Reservoir - Roanoke Arm
Redstone		East Twin - 1974	Brownie
		West Twin - 1974	Mendota
		Virginia	Washington - 1964
		Dutch Hollow	Twin Valley
		West Twin - 1972	Redstone
		Isles	Blackhawk
		Potomac Estuary - Middle Reach	Cox Hollow
		Potomac Estuary - Upper Reach	Virginia
			Isles
			Stewart
			Sallie
			Potomac Estuary - Middle Reach
			Potomac Estuary - Upper Reach

A plot of the ratio of the current phosphorus loading/permissible phosphorus loading and the mean chlorophyll a concentrations for the US OECD water bodies is presented in Figure 86. This correlation was developed to relate the excess phosphorus loading of a water body (as related to its permissible phosphorus loading) to water quality parameters. Lines corresponding to Vollenweider's permissible and excessive loadings (Figure 19) can be inserted in Figure 86. The permissible line corresponds to a current phosphorus load/permissible phosphorus load quotient of one (i.e., the current and permissible phosphorus loads are identical) while a quotient of two (i.e., the current phosphorus load is twice the permissible phosphorus load) denotes the excessive loading level on the Vollenweider diagram. Figure 86 indicates a reasonably good agreement between the investigator-indicated trophic conditions and the predicted trophic conditions based on this relationship. There are apparent anomalies and data scatter which may be due to possible errors in the estimates of either the phosphorus load or mean chlorophyll a values, as well as a number of other factors. For example, the possibility of underestimations of the phosphorus loads for Lakes Dogfish, Lamb and Meander was addressed earlier. The situation with respect to the lag time between a phosphorus loading reduction and a new steady state chlorophyll a concentration for Lakes Washington and Minnetonka have also been addressed. Lake Weir, possibly because of its subtropical nature relative to the northern US temperate conditions of the other US OECD water bodies, also exhibits an anomalous fit.
In general, however, there is a relationship between the current phosphorus load/permissible phosphorus load quotients and the resultant summer chlorophyll a concentrations for the US OECD water bodies. The agreement lends support to the use of this approach for assessing the trophic conditions of water bodies, based on their excess phosphorus loadings above a permissible level, and the resultant chlorophyll a concentrations in the water bodies.

A plot was also made of the ratio of the current phosphorus load to the permissible phosphorus load and the ratio of the current chlorophyll a concentration to the permissible chlorophyll a concentration (Figure 87). As indicated earlier, the permissible chlorophyll a concentration (i.e., 2 µg/l) was the summer mean concentration corresponding to Vollenweider's permissible phosphorus loading line (Figure 19). One can view this graph as a correlation between the "excess" phosphorus loading, as expressed in the current load/permissible load quotient, and the "excess" summer chlorophyll a concentration, as expressed in the current chlorophyll a concentration/summer permissible chlorophyll a concentration (i.e., 2 µg/l) quotient. There is a reasonably good agreement between these two parameters. If the water bodies which have been accepted as anomalous on the basis of various previous analyses (i.e., Lakes Weir, Dogfish, Lamb, Meander, Minnetonka - 1973, Twin Lakes, etc.) are removed, there is a

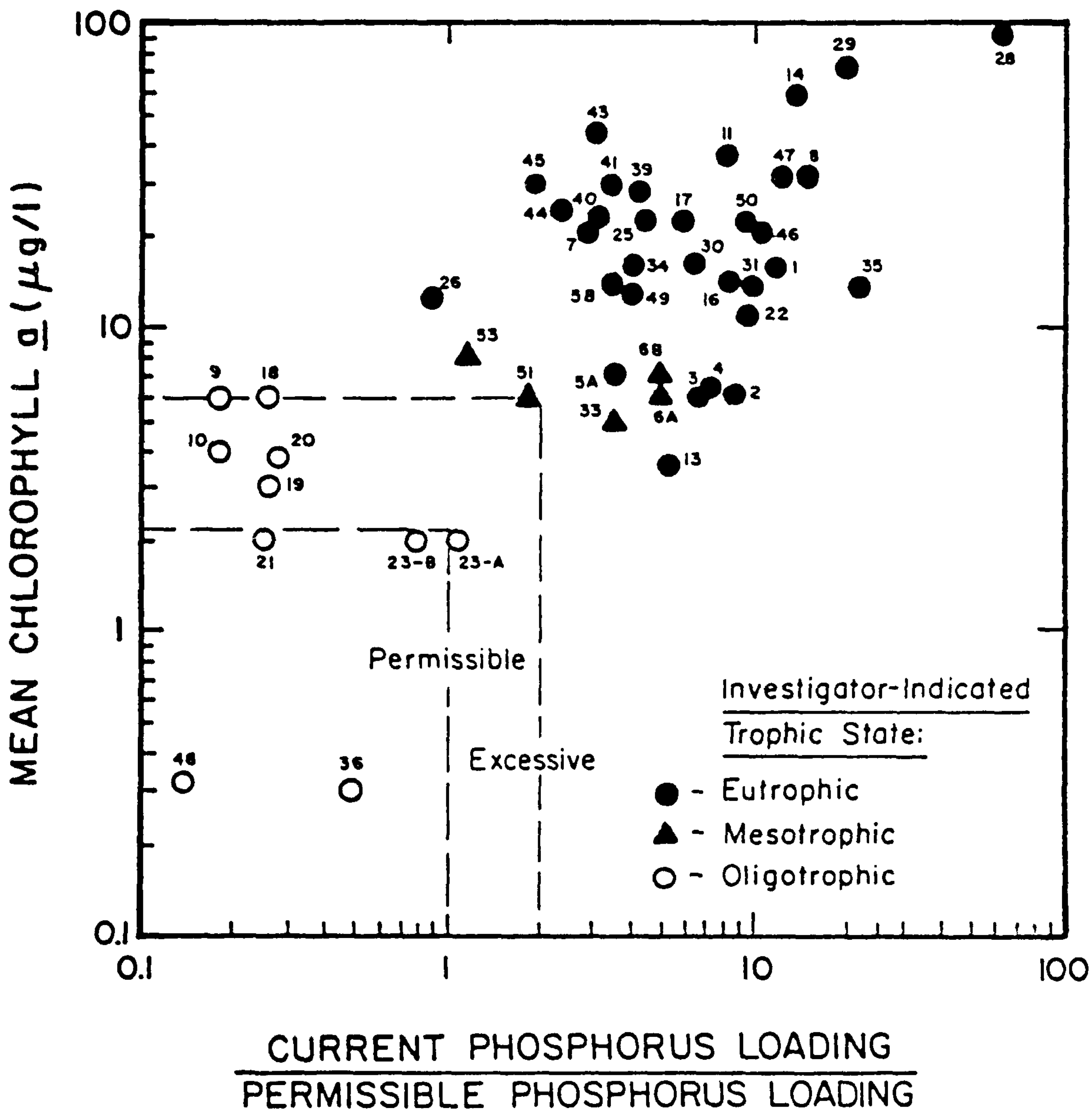

Figure 86. Relationship Between Excessive Phosphorus Loads and Chlorophyll a in US OECD Water Bodies

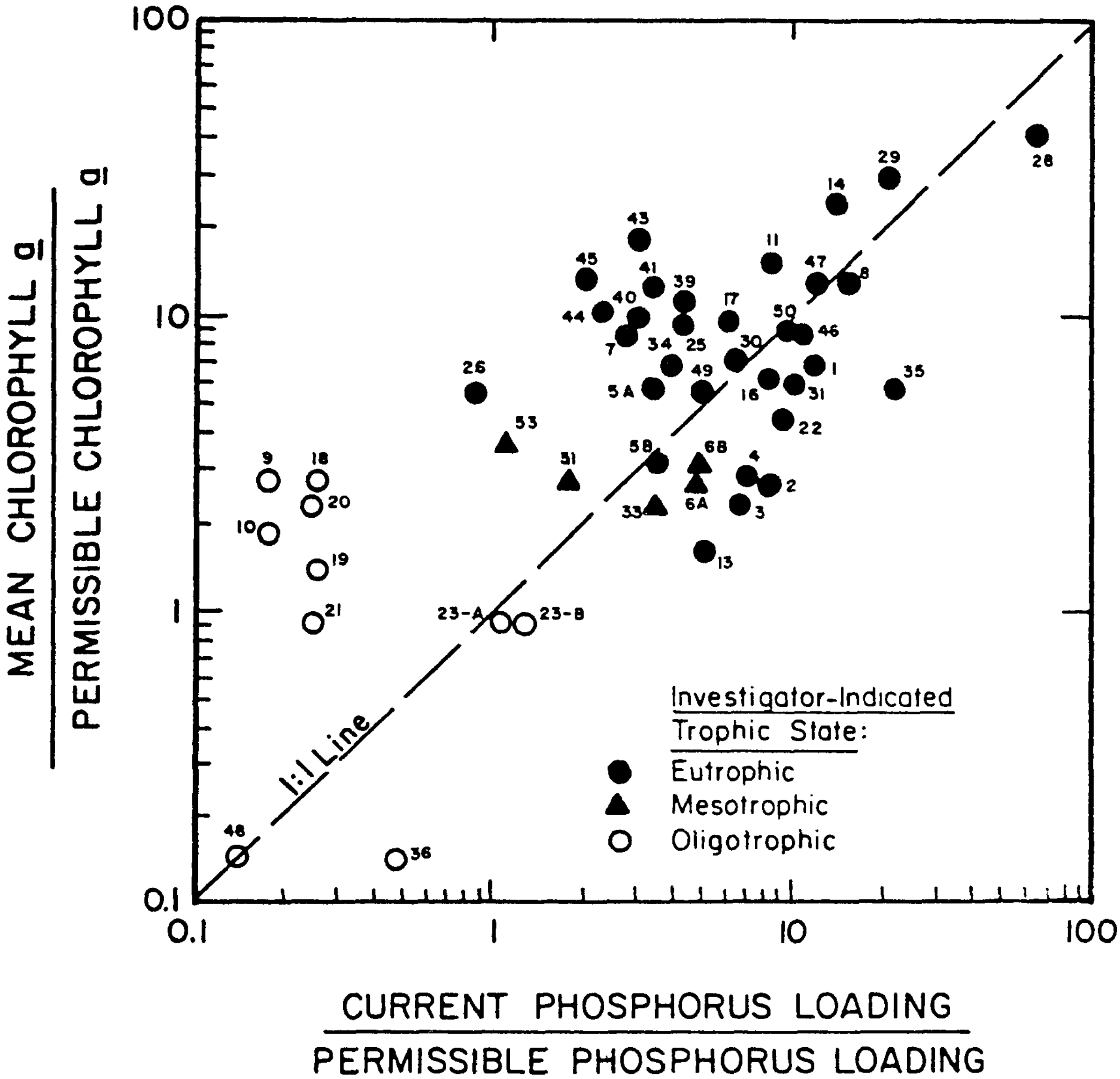

Figure 87. Relationship Between Excessive Phosphorus Loads and Excessive Chlorophyll a in US OECD Water Bodies

better fit of the data sets to a 1:1 relationship. This figure suggests that for a given increase in phosphorus loading to a water body, one can expect a proportional increase in the chlorophyll a concentration. It should be noted that the correlation is a reasonably good one even though the summer chlorophyll a concentrations were not available for all US OECD water bodies, in which case the annual mean values were used in Figure 87. If one accepts a 1:1 relationship between these two parameters, this approach represents a good water quality management tool in that it illustrates that if a water body is receiving three times its permissible phosphorus loading, it can be expected to have a mean epilimnetic summer chlorophyll a of about 3 times the permissible level of 2 µg/l. One can, of course, also use the value of the current phosphorus load/permissible phosphorus load quotient as a trophic ranking system for a wide range of water bodies.

The permissible and excessive phosphorus loading lines on the Vollenweider phosphorus loading diagram (Figure 19) have been related to the water quality parameters of mean summer epilimnetic chlorophyll a, mean Secchi depth, and hypolimnetic oxygen depletion in Figure 88. The basis for this approach was presented earlier. A sequence of increasing chlorophyll a concentrations has been inserted into Figure 88 to illustrate how a variety of boundary loading conditions can be translated into a water quality parameter on Vollenweider's loading diagram. Thus, an individual can literally set his own boundary phosphorus loading levels, as a function of the mean depth/hydraulic residence time characteristics of a water body, based on his own concepts of acceptable chlorophyll a levels during the summer season. Further, by use of Equation 39, which relates chlorophyll a levels and Secchi depths in natural waters, one can also substitute expected Secchi depths as boundary conditions in Vollenweider's loading diagram. Using Equation 39, the permissible loading line (i.e., chlorophyll a concentration of 2 µg/l) corresponds to a Secchi depth of approximately 4.6 meters while the excessive loading line (i.e., chlorophyll a concentration of 6 µg/l) corresponds to 2.7 meters. Finally, with the use of Equation 41, relating hypolimnetic oxygen depletion to Secchi depth, the permissible and excessive loading lines correspond to hypolimnetic oxygen depletion rates of 0.3 and 0.6 mg O_2/m^2/day. These depletion rates can, in turn, be applied to a water body's total hypolimnetic oxygen volume to assess the effects of the phosphorus load on the hypolimnetic oxygen content. These levels are consistent with generally accepted limnological observations.

Thus, this new relationship (Figure 88) appears to relate a phosphorus loading level to the more readily appreciated water quality parameters of chlorophyll a concentrations and Secchi depth. Obviously, it may also be used as a trophic ranking system, based on a water body's predicted chlorophyll a concentrations and/or Secchi depths. It has the feature of relating

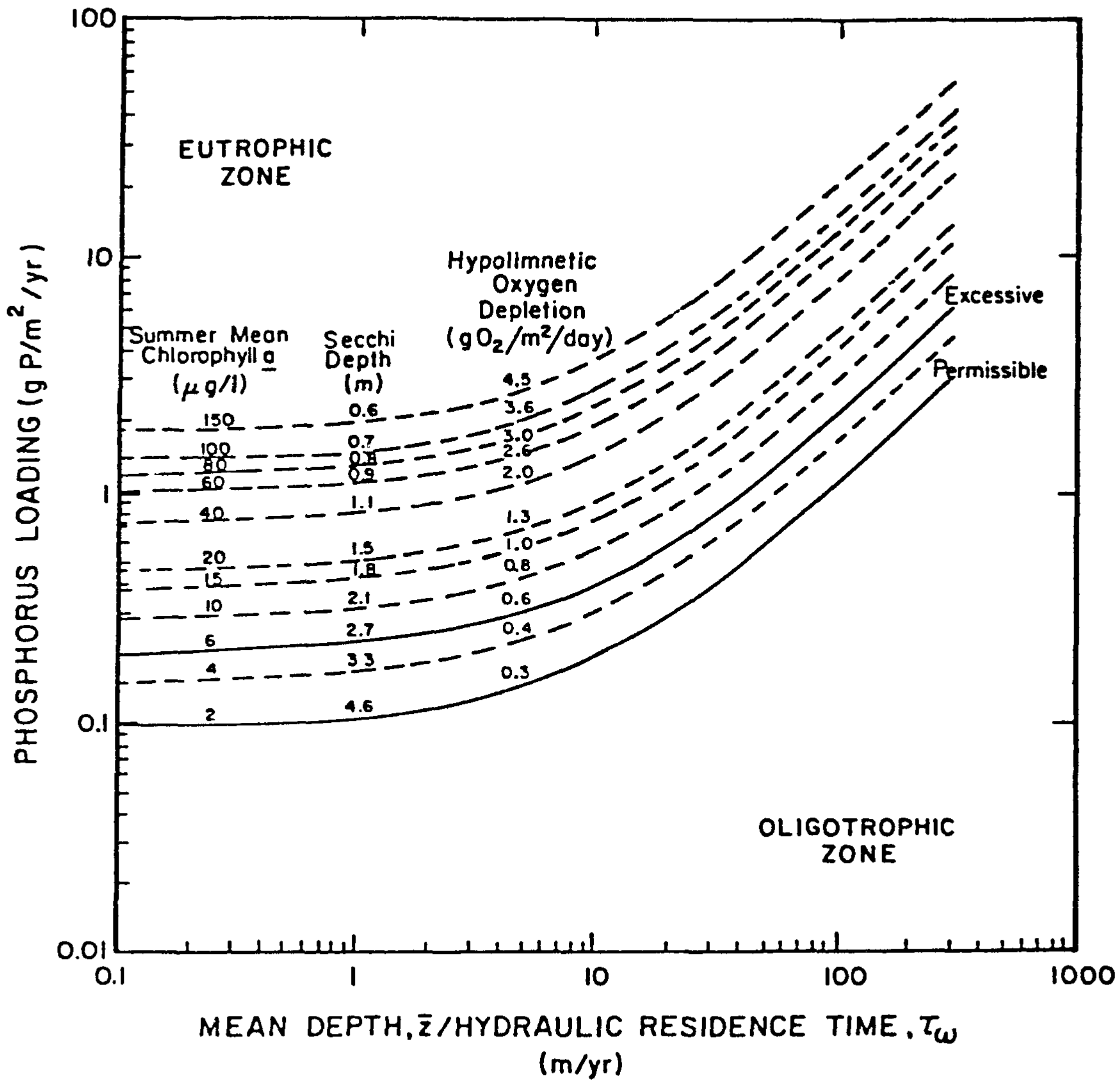

Figure 88. Relationship Between Vollenweider Phosphorus Loading Diagram, Summer Mean Chlorophyll a and Secchi Depth.

Vollenweider's criteria to mean summer conditions in a water body. As indicated earlier, the summer period is usually the period of greatest recreational use of a water body. Consequently, this approach allows individuals concerned with water quality management to predict the phosphorus loading reduction necessary to achieve an "acceptable" summer recreational level of chlorophyll a or transparency in a water body. This can then be translated into costs, using methods indicated in an earlier section, so that the cost-effectiveness of a given eutrophication control program can be evaluated.

It was not possible to satisfactorily test this latter relationship because of lack of sufficient data for the mean summer chlorophyll a concentrations in most of the US OECD water bodies. Even with the data supplied by Dillon and Rigler (1974a) and by Jones and Bachmann (1976), there was still insufficient data for rigorous testing purposes. The data supplied by Dillon and Rigler fit Figure 88 reasonably well, although essentially all his data sets were from oligotrophic water bodies in southern Ontario. The data from Jones and Bachmann on 16 Iowa Lakes (1976) produced a poor fit in Figure 88. However, it was also noted that the data presented by these latter investigators produced poor agreement between predicted and measured chlorophyll a concentrations, by $\pm$ 100 percent in some cases. Jones and Bachmann supplemented their data with literature values for 143 lakes in the determination of their regression equation. However, this data was not presented in their report, and thus could not be tested for its fit in Equation 51. Consequently, the authors of this report offer this model only as a theoretical contribution at the present. However, it has its basis in the same theory and assumptions as does Vollenweider's input-output phosphorus mass balance model (Vollenweider, 1975a; 1976a) and is related to several good correlations between the mean phosphorus, chlorophyll a and Secchi depth in natural waters. This model will be further tested as more data sets become available, and the results reported at a later date. It appears to offer promise as a quantitative tool both for ranking water bodies on a relative trophic scale and for relating phosphorus loads to several readily-appreciated water quality parameters.

In summary, the approaches developed in this study offer methods for the trophic rankings of water bodies based on their displacement from the permissible loading line on the Vollenweider diagram (Figure 19) as related to their predicted summer chlorophyll a and/or Secchi depth characteristics. In general, these approaches appear to complement each other and produce relatively similar results.

SECTION XIII

DISCUSSION

From an overall point of view, based on initial analysis of the US OECD eutrophication study data, it appears that the approach originally developed by Vollenweider and subsequently modified by him, as well as by Dillon and by Larsen and Mercier, has considerable validity as a tool for estimating phosphorus loadings, average phosphorus concentrations and the associated overall degree of fertility for many US lakes and impoundments. In general, based on the US OECD investigators' classifications of the trophic states of their respective water bodies, the US OECD water bodies can be classified into groups with similar phosphorus loads and morphometric and hydrologic characteristics. That is, lakes and impoundments which are generally recognized as being eutrophic in character plot together in each of the various loading-response relationships which have been investigated in this study. While the relationship among the water bodies within a particular group change, depending on the particular relationship being used, the overall relative positions of the water bodies hold reasonably well.

This finding gives considerable validity to the nutrient loading-water body fertility relationship approach originally proposed by Vollenweider and recently adopted by the US EPA as a basis for phosphorus loading water quality criteria (US EPA, 1975b; 1976a). At this time, it appears that the phosphorus loading criteria presented in the US EPA Quality Criteria for Water (US EPA, 1976a) should be modified to include some of the recent modifications of Vollenweider, Dillon and Larsen and Mercier. These modifications are important for water bodies with short hydraulic residence times, such as some impoundments. From the information available today, it appears that water bodies with short hydraulic residence times may have a higher nutrient loading without the same degree of excessive fertilization problems as would be expected in water bodies with longer hydraulic residence times. Conceptually, the nutrients are not present in the water body for a sufficiently long period of time before being flushed out so as to allow their utilization by the aquatic plant populations.

One of the major difficulties that may be encountered in attempting to utilize the US OECD results as a basis for developing uniform national nutrient loading criteria is the fact that, except for one seepage lake in Florida and an impoundment in North Carolina, all of the rest of the US OECD water bodies are located in the colder climates of the East and West coasts and the upper Midwest area of the US. It is possible that nutrient loading-response relationships for water bodies from "typically cold" climates will not hold for the warmer climatic conditions that prevail in the southeast and southwestern US. Additional studies should be conducted on warm water body nutrient-response relationships to ascertain whether these relationships for cold climates are also applicable for warm climates.

Another factor which may play an important role in causing southern water bodies to behave differently from their northern counterparts is that many of these water bodies tend to be more turbid because of suspended sediments resulting from erosion in the watershed and suspension of sediment from the bottom (Lee, 1974b). Some Texas impoundments tend to have severe water quality problems which are associated with floating macrophytes rather than with the planktonic or attached algae typical of excessively fertile waters in cold climates. There is need for nutrient load-response studies such as those currently being conducted as part of the OECD international eutrophication study for water bodies of this type.

There are several aspects of the Vollenweider phosphorus load-fertility response loading diagrams which should be discussed. First, it is clear that the relatively simple model originally developed by Vollenweider is a useful tool to formulate phosphorus load-response relationships in such a way as to be useful as management tools for excessive fertilization control. For the first time, those concerned with control of eutrophication have a basis for predicting the overall trophic state of a particular water body and the associated water quality that will arise from either an increase or decrease in its current nutrient loadings.

With respect to eutrophication modeling, Vollenweider has demonstrated that nutrient loading, lake morphology (as manifested in mean depth) and hydraulic residence time (i.e., "filling time") are the three primary factors which govern lake fertility. As further work is done with the Vollenweider loading curves as part of the international OECD eutrophication study, it is likely other factors (i.e., color, turbidity, seasonal flushing and mixing regimes and temperature) will be found which will further refine the Vollenweider loading relationships and thereby explain apparent deviations from these relationships.

There are certain conditions that must be met before the Vollenweider loading diagram can be used in management of water quality. The most important of these is that the diagram is only valid as a predictive tool in eutrophication control when the primary control of excessive fertilization is through control of a chemical element such as nitrogen or phosphorus, since the loading diagram was developed for a limiting element. It is not technically valid to utilize a loading diagram based on phosphorus loads for eutrophication control when the key limiting element is nitrogen. This report has focused mainly on phosphorus loading relationships. This is justified on the basis of the water bodies that have been included in the US OECD eutrophication study. The majority are phosphorus-limited with respect to algal growth requirements (Table 9). However, there appear to be large numbers of water bodies in the US which are nitrogen-limited. Yet, if phosphorus loading can be decreased to the extent that phosphorus becomes the limiting element in a water body, then the use of Vollenweider's phosphorus loading relationship becomes valid again. This report also discussed the results obtained in the US OECD eutrophication study for nitrogen load-water body response relationships. As discussed in this report, several techniques are available which can be used to assess the key limiting nutrient in a water body.

Another situation in which the Vollenweider loading diagrams may not be applicable is for water bodies with low light penetration. As discussed above, there is reason to believe that water bodies with high inorganic turbidity may behave anomalously, compared to other US OECD water bodies, with respect to their nutrient load-water body response relationships. Piwoni and Lee (1975) have noted a similar phenomenon for highly-colored waters and lakes located in central Wisconsin.

The Vollenweider loading curve may not be applicable without modification to large impoundments with significant amounts of stratified inter- or underflow which would cause nutrients present in the inflow waters to not interact with, or be available to, aquatic plants located in the euphotic zone of the water body. Under these conditions, it may be necessary to modify the loading relationship to utilize a modified hydraulic residence time which would reflect the lack of mixing of the inflowing waters with the euphotic zone waters.

The Vollenweider loading diagram provides some useful information on potential benefits to be derived from manipulating the limiting nutrient input for a particular water body. In general, the log-log plot means that substantial reductions in the nutrient loads must be made before any significant improvement in water quality would be expected. This was discussed in relation to the possible effects of a detergent phosphate ban on eutrophication and water quality in a hypothetical water body in an earlier section of this report.

The Vollenweider loading diagram allows a comparison to be made of the general trophic status of a particular water body relative to a certain nutrient loading. While it is highly successful in categorizing lakes and impoundments into groups with similar trophic states and water quality, such a diagram should not be used as a basis for classification of a water body's trophic status. One should not state that a lake has a particular trophic state merely because of its position on the Vollenweider loading diagram. Rather, one can only indicate that a water body of a given phosphorus loading and mean depth/hydraulic residence time quotient tends to plot in the same relative area of the Vollenweider diagram as water bodies of similar phosphorus loads and mean depth/hydraulic residence time values.

A logical extension of the Vollenweider loading diagram is the development of a relationship between the position of a water body on this curve, or a modification thereof, and the resultant water quality in the water body in which concern is focused on excessive fertilization problems. Ultimately, it should be possible to make a quantitative estimate of the improvement in water quality that may result in a water body from reduction of the nutrient loading by a certain amount. The phosphorus load-chlorophyll *a*, Secchi depth and hypolimnetic oxygen depletion rate relationships (Figures 22, 79 and 80, respectively) and the Vollenweider phosphorus loading diagram incorporating boundary conditions for chlorophyll *a*, Secchi depth and hypolimnetic oxygen depletion (Figure 88) represent significant steps in this direction. Similar types of relationships should be explored for various other types of nutrient load-water quality type response parameters, such as domestic water supply tastes and odors, shortening of water treatment plant filter runs, etc.

Associated with several of the load-response relationships discussed above are descriptive terms such as "excessive", "permissible", "oligotrophic" or "eutrophic" which can be translated into a certain water quality condition. It is important to emphasize that these narrative terms go back to the work of Sawyer (1947) who established critical nutrient concentrations for approximately 20 south-central Wisconsin lakes. Several individuals, including Vollenweider (1968) have found that for many lakes with ice cover during the winter, Sawyer's original critical phosphorus concentrations can be translated into water quality deterioration which typically manifests itself in increased "greenness" of the water. The "greenness" is roughly related to the chlorophyll content of the water. Chlorophyll *a* values of less than 5 μg/l are considered to be indicative of oligotrophic waters with high water quality. Chlorophyll *a* concentrations of greater than about 10 μg/l are often associated with waters classified as eutrophic and possessing deteriorated water quality for many beneficial uses. Chlorophyll concentrations of 2 and 6 μg/l were found for the Vollenweider diagram (Figure 88) permissible and excessive loading lines in an earlier section.

It is important to note, however, that Sawyer and others have been involved with water bodies in which the primary problem was generally the growth of excessive amounts of planktonic algae, and in which this growth affected water clarity. This is manifested in an increased "greenness" during periods of algal blooms and a decreased Secchi depth. In addition, these water bodies generally have planktonic algal growth limited by the phosphorus content of water. In general, these water bodies are natural lakes with little or no color or turbidity. During periods of little or no algal growth, this water has a high degree of clarity with Secchi depths exceeding 3 to 4 meters. The residents of the area who make use of those lakes which have 'excessive' loadings find the water quality sufficiently impaired during periods of algal blooms to curtail recreational use of these water bodies.

The impairment of recreational use (i.e., boating, swimming and fishing) has been used as the basis for determining what constitutes 'excessive' loadings. Lee (1974b) has discussed a possible lack of application of the Sawyer critical nutrient concentrations for the warmer water bodies of the southern US. He noted these critical concentrations may not produce the same deterioration of water quality in the normally turbid or colored waters found in many southern US impoundments as would be expected in water bodies in the north temperate zones of the US. The public does not perceive the same decrease in water clarity, resulting from a certain magnitude of algal blooms, in normally turbid or colored waters as would be perceived in a normally clear water body. Further, Lee (1974b) discussed the fact that in many parts of the US the public will not perceive deteriorated water quality to the same degree since all the water bodies in some areas of the US normally have essentially the same water quality, in contrast to Wisconsin, Michigan and Minnesota, where there are several thousand small lakes of widely varying water quality. Therefore, it must be concluded that, without further study, one cannot assume that permissible' and 'excessive' loading criteria, or for that matter, oligotrophic versus eutrophic waters, are necessarily translated into the same degree of impairment of recreational use in various parts of the US.

In addition to impairing recreational use of water, the stimulation of algal growth by excessive nutrient loading may also cause significant water quality deterioration in domestic and industrial water supplies. Lee (1971) has discussed the potential effects of excessive fertilization on water supply water quality. The most significant problems are those of taste and odor production associated with materials excreted from the algae and a shortening of the length of filter runs. The permissible and excessive criteria used on the various loading diagrams do not consider the potential effects of the nutrients on water supply water quality. From the point of view of eutrophication control in water supply water bodies, at least for certain types of algae, the excessive loading line in the Vollenweider and other phosphorus loading diagrams may have to be lowered significantly in order to minimize the problems of excessive fertility on water quality in these water bodies.

It is important to emphasize that the concepts of excessive nutrient loading pertain to planktonic algal problems and do not consider the problems of attached algae or attached or floating macrophytes. It is highly probable that the permissible and excessive nutrient loadings would also be different in those water bodies which have a tendency to manifest their excessive nutrient concentrations in the growth of nonplanktonic aquatic plants.

Another aspect that should be considered with respect to the Vollenweider loading diagram's emphasis on permissible and excessive loadings based on recreational impairment is that the critical nutrient loading is the loading that impairs recreational use. For many water bodies, algal growth problems which may affect extensive recreational use of the water body are essentially restricted to the summer months. In general, from the point of view of recreational use, there is little concern about the algal blooms that occur in late fall in association with fall overturn and the transport of hypolimnetic nutrients to the surface waters. Further, algal blooms under the ice, or just after ice-out, are usually of little or no significance to impairment of recreational use of the water body. Therefore, as a potential modification of the Vollenweider loading diagram, it is important to consider the nutrient transport to, and cycling within, a water body in relation to how a particular nutrient loading affects water quality. There will likely be situations where major nutrient loads added in late fall or during the winter period will have little or no effect on the following summer's planktonic algal growths. This is an area that needs additional study to determine the critical nutrient loads that have the greatest impact on the water body's water quality.

Examination of the US OECD water bodies for correlations between their nutrient loads and selected eutrophication response parameters (Table 26) has been useful in some instances, although not for all parameters. A major problem which limits the usefulness of many of the correlations is that standardization of data was not possible in many cases. Data for specific parameters was scarce for many water bodies. Further, as indicated in Table 11, a variety of analytical procedures were used to determine the various chemical, biological and physical parameters of interest in the US OECD eutrophication study. Also, a wide variety of sampling methodologies (Appendix II) were employed by the various US OECD investigators.

This lack of uniform analytical and sampling methodologies was due in part to the nature of the US OECD eutrophication study. As indicated in an earlier section, essentially no new-lake studies were begun in the US portion of the OECD

Eutrophication Project. Rather, in general, lakes which had been studied extensively in the past were selected for inclusion in the US OECD eutrophication study. In many cases, the goals of the previous studies on the US OECD water bodies were different from those of the overall OECD efforts. This factor is a cause of at least part of the problem of standardization of data. This lack of standardization has made direct comparability of data between US OECD water bodies of limited value. Inability to compare eutrophication data between water bodies, as indicated by Vollenweider (1968), was a major impetus to initiation of the current OECD Eutrophication Project.

In general, the results of the correlations have indicated the large majority of US OECD water bodies are phosphorus-limited with respect to algal nutrient requirements. The correlations between the phosphorus loads and the various eutrophication response parameters are usually better than those between nitrogen loads and the same response parameters. The exception is the relationship between phosphorus loads and both annual and growing season dissolved phosphorus in the US OECD water bodies, which shows essentially no correlation. By contrast, there is a good correlation between nitrogen loads and inorganic nitrogen, indicating that the inorganic nitrogen is not being used by the algal populations in proportion to its supply to most of the US OECD water bodies. There is essentially no correlation between either dissolved phosphorus or inorganic nitrogen and mean chlorophyll a. By contrast, a good correlation is seen between total phosphorus and mean chlorophyll a supporting the importance of phosphorus in controlling algal growths in most of the US OECD water bodies. This is consistent with the observations concerning algal-limiting nutrients reported by the US OECD investigators (see Table 9).

It is likely that many of the apparently good correlations observed between nitrogen loadings-concentrations and eutrophication response parameters are coincidental artifacts of the relatively constant N:P loading ratio observed in the US OECD water bodies (see Figures 19 and 21). This was noted earlier by Vollenweider (1968), although he used a slightly higher N:P loading ratio (i.e., 15N:1P (by weight)) in the derivation of his nitrogen loading and mean depth relationship (see Figure 6) than was indicated in Figures 19 and 21 in this report.

Several of these correlations were useful in the derivation of several of the relationships derived to evaluate expected changes in water quality resulting from changes in nutrient loads to the US OECD water bodies. Particularly, the relationship between the phosphorus loads and chlorophyll a, between chlorophyll a and Secchi depth, between hypolimnetic oxygen depletion rate and Secchi depth, and between spring overturn total phosphorus and summer chlorophyll a served as the basis for most of these water quality models (see Figures 22, 78 and 79). These relationships have been observed in many other water bodies, in addition to the US OECD water bodies, substantiating their occurrence in water bodies of differing trophic conditions. Using the water quality model relationships derived in this report, it is now possible to make a technically sound evaluation of the effects of any given water quality management program. In the past, eutrophication control programs have largely been directed toward the removal of phosphorus from domestic wastewater sources. However, this approach has been largely subjective. The water quality models derived in this report offer practical tools for individuals concerned with water quality management and eutrophication control.

These water quality models have several advantages over previous eutrophication modeling efforts. First, they are related to common eutrophication response parameters which are readily discernible to both scientist, engineer and layman. While the Vollenweider loading diagram (Figure 19) offers a good indication of the overall eutrophication of the US OECD water bodies, these water quality models then relate the relative degree of fertility of the US OECD water bodies into three common eutrophication response parameters, namely chlorophyll a concentrations, Secchi depth, and the hypolimnetic oxygen depletion rate. These first two parameters, both related to the "greenness" or transparency of water bodies, are more widely appreciated and understood as a good overall indicator of water quality that the public could perceive than would be the knowledge concerning the extent of areal total phosphorus loading reduction necessary to achieve a permissible phosphorus load. Another feature of these models is that they are simple, requiring only knowledge of easily-measured parameters. They are also based on observations concerning nutrient load-eutrophication response relationships which have been observed in a wide range of water bodies, lending credibility to their general applicability.

One of their main features is that they allow evaluation of the effects of a phosphorus eutrophication control program prior to initiation of the program. This information will enable water quality managers to inform the public of the expected increase in water quality that can be achieved as a result of controlling phosphorus from each of the potentially available sources for a particular water body to a selected degree. A proper cost-benefit analysis can then be conducted for a given eutrophication control program prior to its initiation. With this knowledge and the water quality models derived in this report, the public can then determine whether the expected results of a given eutrophication control program are justified by its expense. Lee (1976) has used these above approaches in evaluating the expected water quality benefits to be derived for the Great Lakes from a phosphate-built detergent ban in the State of Michigan.

The Vollenweider phosphorus loading diagram has been related to the same water quality parameters (see Figure 88). This relationship was derived in an earlier section of this report (i.e., Equations 40, 47 and 48). Vollenweider's later models (1976a), as well as those of Dillon (1975) and Larsen and Mercier (1976) have their basis in the same theoretical phosphorus mass balance approach as was used to derive the Vollenweider phosphorus loading diagram (Figure 19). Consequently, relating the Vollenweider phosphorus loading diagram to these water quality parameters is pleasing in that it relates the above-mentioned models of Vollenweider, Dillon and Larsen and Mercier to these same parameters. Relating them to more readily appreciated water quality parameters will likely enhance their application as eutrophication evaluation methodologies.

The results of the trophic status index study indicated that, in general, the trophic classification systems of the US EPA (1974d), Carlson (1974) and Piwoni and Lee (1975) produce approximately the same relative trophic rankings for the US OECD water bodies. There are a few anomalies noted with all three indices, with some water bodies of more fertile conditions ranked more toward the oligotrophic end of the trophic spectrum than less fertile water bodies. All three ranking systems producing similar results may be partially due to the fact that all three systems have several common parameters. These parameters may have been of sufficient importance in the trophic rankings, relative to the other parameters, that they influenced all three systems toward similar results.

The approach developed in this report of ranking the US OECD water bodies on the basis of their excess phosphorus loads (i.e., ratio of current phosphorus load to permissible phosphorus load) offers another simple method of relating phosphorus loads to eutrophication response parameters. Examination of the relationship between the excess phosphorus load and mean chlorophyll a (Figure 86) shows a positive correlation exists between these parameters, although there is a scatter of the data. This data scatter is due in part to the fact that the mean chlorophyll a values used in this relationship are a mixture of annual and growing season values. This relationship is similar to that of Vollenweider which relates the phosphorus load, as modified by mean depth and hydraulic residence time, to the mean chlorophyll a (see Figure 22), except that the chlorophyll a is being correlated with the excess phosphorus load in this model.

The relationship between excess chlorophyll a and excess phosphorus load also showed good promise as a water body trophic ranking system. The excess chlorophyll a was referenced to the permissible 2 µg/l chlorophyll a concentration derived in an earlier section (see Equations 48 and 49). This relationship (Figure 87) is interesting in that, although the data is somewhat scattered, it appears to illustrate a 1:1 relationship between the excess phosphorus load and excess chlorophyll a in the US OECD water bodies. This suggests that a water body receiving a phosphorus load of a certain magnitude above the permissible level will experience a mean summer chlorophyll a level of essentially the same proportion above the 'permissible' chlorophyll level. While this is not unexpected to some degree, it is surprising to note that this approximately 1:1 relationship between excess phosphorus and excess chlorophyll a appeared to hold over the whole phosphorus loading and chlorophyll a range of the US OECD water bodies. Thus, one could use the current phosphorus load/permissible phosphorus load quotient as a trophic ranking system for a wide range of water bodies.

The applicability of the Vollenweider loading relationships for shallow lakes is an area that needs further attention. Examination of the US OECD eutrophication study data, although limited for these types of water bodies, shows that shallow lakes and impoundments do not appear to have significantly different chlorophyll a and Secchi depth responses to phosphorus loads than do the other US OECD water bodies (Figures 22 and 79, respectively). It should be noted that the nutrient load estimates for many of the shallow lakes and impoundments are based on land use in the watershed and the appropriate nutrient export coefficients. Because of the uncertainty of the nutrient loads for these water bodies at this time, it would be inappropriate to conclude that shallow water bodies have different nutrient load-eutrophication response relationships than do deeper water bodies.

The primary distinguishing feature between shallow lakes and deeper lakes is the absence of thermal stratification. For the purposes of this report, a shallow lake is one with a mean depth of 3 m or less. Generally, water bodies of this type do not thermally stratify, except under highly sheltered conditions in which wind-induced mixing of the water column is hampered. The lack of permanent thermal stratification during the growing season plays a major role in nutrient recycling. In deep lakes (i.e., lakes that remain thermally-stratified during the entire growing season), the thermocline represents a barrier to nutrient recycling from the hypolimnetic waters. The effectiveness of the thermocline as a nutrient barrier is highly variable and varies from lake to lake. As discussed by Stauffer and Lee (1973), some water bodies, such as Lake Mendota in Madison, Wisconsin, which permanently stratify during the summer, still derive appreciable nutrients from the hypolimnion, as a result of thermocline migration. In fact, this phenomenon appears to be the primary controlling mechanism governing many of the algal blooms that occur in Lake Mendota during the summer.

As shown by Lee et al (1976), appreciable phosphorus recycling occurs in aerobic waters. This recycling is associated primarily with mineralization of algal phosphorus. This phenomenon would be especially important in shallow lakes because they tend to have warmer water overall than the surface waters of deeper lakes of the same region. The higher temperatures in the shallow lakes would promote the phosphorus mineralization. This higher overall temperature, in addition to increasing the rate of nutrient recycling, also affects many other factors controlling algal growth including the algal growth rate and algal predation by zooplankton. Further, higher temperatures would likely have some effects on the types of algae present. It is therefore reasonable to conclude that, as a result of their somewhat elevated temperatures compared to deeper water bodies, shallow lakes would tend to use their nutrients, especially phosphorus, to a somewhat greater degree and at a faster rate. This could cause shallow lakes to not fit as well as deeper water bodies in the Vollenweider nutrient load-eutrophication response relationships or to deviate from the Vollenweider nutrient load relationships which were developed for deeper water bodies.

Another factor which could influence the behavior of shallow lakes, compared to deeper water bodies, in the Vollenweider phosphorus loading relationships is water clarity. In general, shallow water bodies tend to be more turbid as a result of suspension of the sediments into the water column. This suspension arises from several factors, the most important of which is wind-induced mixing. Also important in their suspension is the mixing of sediments to the overlying waters from the activities of fish, such as carp burrowing in the sediments. As discussed by Lee (1970), anaerobic fermentation of the sediments, as well as benthic organism biomass suspension due to photosynthesis, also contribute to the mixing of the sediments in the water column. Another factor which would tend to make shallow lakes more turbid in hardwater areas is the precipitation of calcium carbonate which, under certain extreme conditions, can produce a "milky" appearance in the water column.

The elevated turbidity often present in shallow lakes could cause these water bodies to deviate from the Vollenweider relationships in a variety of ways. One of the most important of these possible deviations is the promotion of light limitation of algal growth. Therefore, even though water temperatures would tend to be higher and aerobic nutrient recycling faster in shallow lakes, algal growth in these water bodies may not be stimulated because of increased detrital and mineral activity in the water, which could cause a light limitation of algal growth in these water bodies.

This increase in nonalgal turbidity in shallow lakes would tend to make phosphorus somewhat less available for algal growth because of sorption and precipitation reactions in the water body. Detrital minerals, especially clays, have a relatively high capacity for phosphate uptale. Also, calcium carbonate precipitation in hard water systems would probably result in coprecipitation of hydroxyapatites. On the other hand, since the water in shallow lakes is almost always oxygenated, phosphate sorption by freshly precipitated iron hydroxide would be minimal. Thus, from an overall point of view, it is likely that less of the phosphorus added to a shallow lake would be available to promote algal growth than would be seen in deeper water bodies.

The increased turbidity often present in shallow lakes would tend to greatly alter the public's response to planktonic algal growths. The public in general tends to perceive change in a water body as a significant detrimental factor. Planktonic algal growth in a water body that is generally somewhat turbid because of sediment suspension in the water column would be less objectionable to the public since the effect of the algal on overall water clarity is more difficult to perceive than in less turbid water bodies. In a study currently being conducted by Lee et al. (1977), it has been found that Lake Ray Hubbard, an impoundment near Dallas, Texas, tends to have a markedly different chlorophyll-Secchi depth relationship than do the US OECD water bodies. Several arms of this impoundment are 1 to 3 m deep and contain large amounts of mineral and detrital turbidity in the water column. A given planktonic algal chlorophyll in this lake is associated with a significantly shallower Secchi depth than found in typical US OECD eutrophication study water bodies. Large algal blooms occur in this lake, yet have limited impact on its recreational use because the planktonic chlorophyll does not change overall water clarity to a significant degree compared to non-bloom conditions in the water body.

Many shallow lakes and the shallow waters of deeper lakes tend to support large populations of attached algae and macrophytes. Since the Vollenweider nutrient relationships are based primarily on planktonic algal chlorophyll, growth of non-planktonic plants tend to act as a sink for nutrients during the growing season. Therefore, less planktonic algal production will occur in shallow lakes containing high populations of attached algae and macrophytes.

From the above discussion it is apparent that a variety of factors would tend to cause shallow lakes to deviate from the Vollenweider nutrient load-eutrophication response relationships. However, the effects of many of these factors tend to oppose one another, with the result that it is impossible at this time to predict, without additional study, whether shallow lakes and impoundments will tend to show different nutrient load-eutrophication response relationships than other deeper water bodies. The combined OECD Eutrophication Program study data from the Alpine, Nordic, North American and Shallow Lakes and Impoundments Projects will likely provide a sufficient data base to determine whether shallow lakes and impoundments tend to deviate significantly from the nutrient load-eutrophication response relationships than deeper water bodies.

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APPENDIX I

FINAL REPORT OUTLINE
(North American Project)

I. Introduction - Short Past History of Water Body

II. Brief Geographical Description of Water Body
   A. Latitude and Longitude (Centroid of Water Body)
   B. Altitude Above or Below Sea Level
   C. Catchment Area (Including Area of Surface Water)
   D. General Climatic Data (Ice Coverage; Average Monthly Air Temperature; Wind Patterns; Evaporation; etc.)
   E. General Geological Characteristics (Nature of Bedrock; Subsoil and Soils; Importance of Land Erosion)
   F. Vegetation
   G. Population
   H. Land Usage (Industrial, Urban, Agricultural, etc.)
   I. Use of Water (Drinking, Sport, Fishing, etc.)
   J. Wastewater Discharges (Population and Industry)

III. Brief Description of Morphometric and Hydrologic Characteristics of Water Body
   A. Surface Area of Water (Length, Width, Shore Length, etc.)
   B. Volume of Water (Information on Regulation)
   C. Maximum and Mean Depth
   D. Ratio of Epilimnion over Hypolimnion
   E. Duration of Stratification
   F. Nature of Lake Sediments
   G. Seasonal Variation of Monthly Precipitation (Maximum, Minimum Conditions on Drainage Basin)

H. Inflow and Outflow of Water (Also Underground)

I. Water Currents

J. Water Renewal Time (Residence Time)

IV. Limnological Characterization Summary

A. Physical

1. Temperature
2. Conductivity
3. Light
4. Color
5. Solar Radiation

B. Chemical

1. pH
2. Dissolved Oxygen
3. Total Phosphorus (Including Fraction Forms)
4. Total Nitrogen (Including Fraction Forms)
5. Alkalinity and/or Acidity
6. Ca, Mg, Na, K, SO_4, Fe

C. Biological

1. Phytoplankton (Chlorophyll; Primary Productivity; Algal Assays; Identification and Count)
2. Zooplankton (Identification and Count)
3. Bottom Fauna
4. Fish
5. Bacteria
6. Bottom Flora
7. Macrophytes

V. Nutrient Budgets Summary

A. Phosphorus

Source	Kg/Yr
Waste Discharges	xx
Land Runoff	xx
Precipitation	xx
Ground Water	xx
Other	xx
Total	xx

B. Nitrogen

Source	Kg/Yr
Waste Discharges	xx
Land Runoff	xx
Precipitation	xx
Ground Water	xx
Other	xx
Total	xx

C. Other Nutrient Budgets, If Available

VI. Discussion

A. Limnological Characterization

B. Delineation of Trophic Status

C. Trophic Status Versus Nutrient Budgets

1. Present Vollenweider Numbers (Grams/Meter^2/Year)

2. Mean Depth/Hydraulic Residence Time

VII. Summary

BLACKHAWK (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	3.6×10^7 m^2
Water Body Surface Area	8.9×10^5 m^2
Mean Depth	4.9 m
Hydraulic Residence Time	0.5 yr
Mean Alkalinity	227 mg/l as $CaCO_3$
Mean Conductivity	471 μmhos/cm @ 25°C
Mean Secchi Depth[b]	3.6 m
Mean Dissolved Phosphorus[b]	0.04[c] mg P/l
Mean Total Phosphorus[b]	0.12[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	1.02[c] mg N/l
Mean Chlorophyll a[b]	14.6 μg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	1900-2070 kg P/yr
Surface Area Loading	2.13-2.32 g P/m^2/yr
Nitrogen Loading:	
Point Source	0 kg P/yr
Non-Point Source	20,900 kg N/yr
Surface Area Loading	23.4 g N/m^2/yr
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.05 mg P/l
Dissolved Phosphorus	0.015 mg P/l
Inorganic Nitrogen	0.54 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six-week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentrations.

Dash (-) indicates no data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

BROWNIE (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1971
Drainage Area[a]	4.7×10^5 m^2
Water Body Surface Area	7.3×10^4 m^2
Mean Depth	6.8 m
Hydraulic Residence Time	2.0 yr
Mean Alkalinity	123-136 mg/l as $CaCO_3$
Mean Conductivity	400-475 μmhos/cm @ 25°C
Mean Secchi Depth	1.5 m
Mean Dissolved Phosphorus	< 0.01[b] mg P/l
Mean Total Phosphorus	-
Mean Inorganic Nitrogen (NH_4^++NO_3^- as N)	< 0.055[b] mg N/l
Mean Chlorophyll a	5.9[b] μg/l
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	82.1[c] kg P/yr
Non-Point Source	3.8 kg P/yr
Surface Area Loading	1.18 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	-
Non-Point Source	-
Surface Area Loading	-

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Summer average surface values

[c]Includes urban storm water drainage.

Dash (-) indicates no data available.

*Data taken from Shapiro (1975a) and personal communication (Table 3).

CALHOUN (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1971
Drainage Area[a]	7.6×10^6 m^2
Water Body Surface Area	1.7×10^6 m^2
Mean Depth	10.6 m
Hydraulic Residence Time	3.6 yr
Mean Alkalinity	80-123 mg/l as $CaCO_3$
Mean Conductivity	400-500 µmhos/cm @ 25°C
Mean Secchi Depth	2.1 m
Mean Dissolved Phosphorus	< 0.005[b] mg P/l
Mean Total Phosphorus	0.106[b] mg P/l
Mean Inorganic Nitrogen ($NH_4^+ + NO_3^-$ as N)	< 0.055[b] mg N/l
Mean Chlorophyll a	6.0[b] µg/l
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	1370[c] kg P/yr
Non-Point Source	91 kg P/yr
Surface Area Loading	0.86 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	-
Non-Point Source	-
Surface Area Loading	-

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Summer surface average values.

[c]Includes urban stormwater drainage.

Dash (-) indicates no data available.

*Data taken from Shapiro (1975a) and personal communication (Table 3).

CAMELOT-SHERWOOD (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	9.1×10^7 m^2
Water Body Surface Area	2.8×10^6 m^2
Mean Depth	3 m
Hydraulic Residence Time	0.09 - 0.14 yr
Mean Alkalinity	125 mg/l as $CaCO_3$
Mean Conductivity	311 μmhos/cm @ 25°C
Mean Secchi Depth[b]	2.0 m
Mean Dissolved Phosphorus[b]	0.008[c] mg P/l
Mean Total Phosphorus[b]	0.03[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	1.07[c] mg N/l
Mean Chlorophyll _a_ [b]	6.3 μg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	6600-7580 kg P/yr
Surface Area Loading	2.35-2.68 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	97,600 kg N/yr
Surface Area Loading	34.6 g $N/m^2/yr$
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.04 mg P/l
Dissolved Phosphorus	0.008 mg P/l
Inorganic Nitrogen	0.59 mg N/l

Investigator-Indicated Comments

Lake highly colored because of humic content.

[a]Does not include water body surface area.

[b]Data based on samples obtained at six-week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentrations.

Dash (-) indicates no data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

CANADARAGO (N.Y.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	[a]Eutrophic in 1968-1969	
Drainage Area[b]	1.8×10^8 m^2	
Water Body Surface Area	7.6×10^6 m^2	
Mean Depth	7.7 m	
Hydraulic Residence Time	0.6 yr	
Mean Alkalinity	248 mg/l as $CaCO_3$	
Mean Conductivity	223 μmhos/cm @ $25^{o}C$	
Mean Secchi Depth[c]	1.8 m	
	1968	1969
Mean Dissolved Phosphorus[c]	0.02	0.02 mg P/l
Mean Total Phosphorus[c]	0.05	0.04 mg P/l
Mean Inorganic Nitrogen[c] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.38	0.44 mg N/l
Mean Chlorophyll a	13	7 μg/l
Annual Primary Productivity	1971=195; 1972=136; 1973=236 g $C/m^2/yr$	
Phosphorus Loading:		
Point Source	2800 kg P/yr	
Non-Point Source	3200 kg P/yr	
Surface Area Loading	0.8 g $P/m^2/yr$	
Nitrogen Loading:		
Point Source	7800 kg N/yr	
Non-Point Source	128,600 kg N/yr	
Surface Area Loading	18.0 g $N/m^2/yr$	

	Growing Season(May-Sept) Mean Epilimnetic Values		Spring Overturn Values	
	1968	1969	1968	1969
Mean Secchi Depth (m)	-	1.7	-	-
Total Phosphorus (mg P/l)	0.06	0.04	0.02	0.03
Dissolved Phosphorus(mg P/l)	0.020	0.013	0.016	0.015
Inorganic Nitrogen (mg N/l)	0.21	0.30	0.38	0.44
Chlorophyll a (μg/l)	9	5	-	-

Investigator-Indicated Comments

[a]Prior to completion of tertiary waste treatment plant for treatment of major point source nutrient input in 1972.

(continued)

DATA SUMMARY FOR CANADARAGO (N.Y.)* - (continued)

[b]Does not include water body surface area.

[c]Data based on samples obtained monthly from early May-late November, 1968-1969, from ten stations at the 0-4.5 m depth, 4.5-9.0 m depth, and 9.0 bottom depth.

Dash (-) indicates no data available.

*Data taken from Hetling *et al*. (1975) and personal communication (Table 3).

CAYUGA (N.Y.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Mesotrophic in 1972-1973	
Drainage Area[a]	2.0×10^9 m^2	
Water Body Surface Area	1.7×10^8 m^2	
Mean Depth	54 m	
Hydraulic Residence Time	8.6 yr	
Mean Alkalinity	102 mg/l as $CaCO_3$	
Mean Conductivity	575 μmhos/cm @ 25°C	
	1972	1973
Mean Secchi Depth[b]	2.3	2.3 m
Mean Dissolved Phosphorus[b]	0.003	0.004 mg P/l
Mean Total Phosphorus[b]	0.02	0.02 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.37	0.51 mg N/l
Mean Chlorophyll *a*[b]	6	5 μg/l
Annual Primary Productivity	58[c] g C/m^2/yr	
Phosphorus Loading[d]:		
Point Source	63,900 kg P/yr	
Non-Point Source	77,100 kg P/yr	
Surface Area Loading	0.8 g P/m^2/yr	
Nitrogen Loading[d]:		
Point Source	168,000 kg N/yr	
Non-Point Source	2,300,000 kg N/yr	
Surface Area Loading	14.3 g N/m^2/yr	

	Growing Season (May-Sept) Mean Epilimnetic Values	
	1972	1973
Secchi Depth (m)	1.8	2.4
Dissolved Phosphorus (mg P/l)	0.003	0.001
Inorganic Nitrogen (mg N/l)	0.35	0.36
Chlorophyll *a* (μg/l)	7.4	5.6

Total Phosphorus ranges from 0.015-0.022 mg/l throughout water column during all seasons of the year.

Investigator-Indicated Comments

[a]Data does not include water body surface area.

(continued)

[b]Data based on samples collected at three-five sampling stations in 1972-1973, at surface, 2m, 5m and 10m, at weekly intervals during June-August, biweekly intervals during mid-April-May and September-October, and monthly intervals the rest of the year, down the long axis of the lake.

[c]Based on Barlow (1969) and Peterson (1971).

[d]1970-1971 data.

*Data taken from Oglesby (1975) and personal communication (Table 3).

CEDAR (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1971
Drainage Area[a]	1.6×10^6 m^2
Water Body Surface Area	6.9×10^5 m^2
Mean Depth	6.1 m
Hydraulic Residence Time	3.3 yr
Mean Alkalinity	71-109 mg/l as $CaCO_3$
Mean Conductivity	400 μmhos/cm @ 25°C
Mean Secchi Depth	1.8 m
Mean Dissolved Phosphorus	< 0.005[b] mg P/l
Mean Total Phosphorus	0.055[b] mg P/l
Mean Inorganic Nitrogen (NH_4^++NO_3^- as N)	< 0.055[b] mg N/l
Mean Chlorophyll a	20[b] μg/l
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	205[c] kg P/yr
Non-Point Source	36 kg P/yr
Surface Area Loading	0.35 g P/m^2/yr
Nitrogen Loading:	
Point Source	-
Non-Point Source	-
Surface Area Loading	-

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Summer surface average values.

[c]Includes urban stormwater drainage.

Dash (-) indicates no data available.

*Data taken from Shapiro (1975a) and personal communication (Table 3).

COX HOLLOW LAKE (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	1.6×10^7 m^2
Water Body Surface Area	3.9×10^5 m^2
Mean Depth	3.8 m
Hydraulic Residence Time	0.5 - 0.7 yr
Mean Alkalinity	205 mg/l as $CaCO_3$
Mean Conductivity	440 μmhos/cm @ 25°C
Mean Secchi Depth[b]	1.5 m
Mean Dissolved Phosphorus[b]	0.04[c] mg P/l
Mean Total Phosphorus[b]	0.10[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.83[c] mg N/l
Mean Chlorophyll *a*[b]	26.5 μg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	630-810 kg P/yr
Surface Area Loading	1.62-2.08 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	7410 kg N/yr
Surface Area Loading	19.1 g $N/m^2/yr$
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.06 mg P/l
Dissolved Phosphorus	0.02 mg P/l
Inorganic Nitrogen	0.36 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six-week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentration.

Dash (-) indicates no data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

DOGFISH (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Oligotrophic in 1971-1972	
Drainage Area[a]	8.8×10^5 m^2	
Water Body Surface Area	2.9×10^5 m^2	
Mean Depth	4.0 m	
Hydraulic Residence Time	3.5 yr	
	1971	1972
Mean Alkalinity[b]	8	10 mg/l as $CaCO_3$
Mean Conductivity[b]	17.3	16.0 µmhos/cm @ 25°C
Mean Secchi Depth[b]	2.7	2.5 m
Mean Dissolved Phosphorus	-	-
Mean Total Phosphorus[b]	0.010	0.010 mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+NO_2^- as N)	0.39	- mg N/l
Mean Chlorophyll a	6 (4)[c]	4 (2)[c] µg/l
Annual Primary Productivity	-	-
Phosphorus Loading:		
Point Source	-	0 kg P/yr
Non-Point Source	-	4.9 kg P/yr
Surface Area Loading	-	0.02 g $P/m^2/yr$
Nitrogen Loading:		
Point Source	-	-
Non-Point Source	-	-
Surface Area Loading	-	-

Mean pH = 6.0

Investigator-Indicated Comments

Water slightly stained with humics.

Phytoplankton characterized by chrysophytes and cryptomonads except during summer and fall, when greens and blue greens were significant.

[a]Does not include water body surface area.

[b]May-October mean values for 1971-1972.

[c]Euphotic zone values.

Dash (-) indicates no data available.

*Data taken from Tarapchak et al. (1975) and personal communication (Table 3).

DUTCH HOLLOW LAKE (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	1.2×10^7 m^2
Water Body Surface Area	8.5×10^5 m^2
Mean Depth	3 m
Hydraulic Residence Time	1.8 yr
Mean Alkalinity	133 mg/l as $CaCO_3$
Mean Conductivity[b]	252 μmhos/cm @ 25°C
Mean Secchi Depth[b]	0.8 m
Mean Dissolved Phosphorus[b]	0.020[c] mg P/l
Mean Total Phosphorus[b]	0.40[c] mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+ NO_2 as N)	0.61[c] mg N/l
Mean Chlorophyll a [b]	33.9 μg/l (first 2 meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	810-870 kg P/yr
Surface Area Loading	0.95-1.01 g P/m^2/yr
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	8840 kg N/yr
Surface Area Loading	10.4 g N/m^2/yr
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.12 mg P/l
Dissolved Phosphorus	0.01 mg P/l
Inorganic Nitrogen	0.22 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentrations.

Dash (-) No data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

GEORGE (N.Y.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Oligotrophic-Mesotrophic in 1972-73.
Drainage Area[a]	6.1×10^8 m^2
Water Body Surface Area	1.1×10^8 m^2
Mean Depth	18 m
Hydraulic Residence Time	8 yr
Mean Alkalinity	21 mg/l as $CaCO_3$
Mean Conductivity	86 µmhos/cm @ 25°C
Mean Secchi Depth	8.5 m
Mean Dissolved Phosphorus	0.002 mg P/l
Mean Total Phosphorus	0.0085 mg P/l
Mean Inorganic Nitrogen (NH_4^++NO_3^- as N)	0.05 mg N/l
Mean Chlorophyll a	-
Annual Primary Productivity	7.2 g C/m^2/yr
Phosphorus Loading:	
Point Source	80 kg P/yr
Non-Point Source	7800 kg P/yr
Surface Area Loading	0.07 g P/m^2/yr
Nitrogen Loading:	
Point Source	17,700 kg N/yr
Non-Point Source	201,000 kg N/yr
Surface Area Loading	1.8 g N/m^2/yr

Investigator-Indicated Comments

[a]Does not include water body surface area.

Dash (-) indicates no data available.

*Data taken from Ferris and Clesceri (1975) and personal communication (Table 3).

HARRIET (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1971
Drainage Area[a]	4.8×10^6 m^2
Water Body Surface Area	1.4×10^6 m^2
Mean Depth	8.8 m
Hydraulic Residence Time	2.4 yr
Mean Alkalinity	92 - 124 mg/l as $CaCO_3$
Mean Conductivity	360-425 μmhos/cm @ 25°C
Mean Secchi Depth	2.4 m
Mean Dissolved Phosphorus	<0.005[b] mg P/l
Mean Total Phosphorus	0.062[b] mg P/l
Mean Inorganic Nitrogen (NH_4^++NO_3^- as N)	<0.055[b] mg N/l
Mean Chlorophyll <u>a</u>	3.5[b] μg/l
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	890[c] kg P/yr
Non-Point Source	126 kg P/yr
Surface Area Loading	0.71 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	-
Non-Point Source	-
Surface Area Loading	-

<u>Investigator-Indicated Comments</u>

[a]Does not include water body surface area.

[b]Summer average surface values.

[c]Urban stormwater drainage only.

Dash (-) indicates no data available.

*Data taken from Shapiro (1975a) and personal communication (Table 3).

ISLES (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1971
Drainage Area[a]	2.8×10^6 m^2
Water Body Surface Area	4.2×10^5
Mean Depth	2.7 m
Hydraulic Residence Time	0.6 yr
Mean Alkalinity	68-131 mg/l as $CaCO_3$
Mean Conductivity	380-470 μmhos/cm @ 25°C
Mean Secchi Depth	1.0 m
Mean Dissolved Phosphorus	<0.010[b] mg P/l
Mean Total Phosphorus	0.110[b] mg P/l
Mean Inorganic Nitrogen (NH_4^++NO_3^- as N)	<0.055[b] mg N/l
Mean Chlorophyll _a_	53[b] μg/l
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	828[c] kg P/yr
Non-Point Source	23 kg P/yr
Surface Area Loading	2.03 g P/m^2/yr
Nitrogen Loading:	
Point Source	-
Non-Point Source	-
Surface Area Loading	-

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Summer surface average values.

[c]Urban storm water drainage only.

Dash (-) No data available.

*Data taken from Shapiro (1975a) and personal communication (Table 3).

KERR RESERVOIR (N. CAROLINA-VIR.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic-Mesotrophic in 1975	
Drainage Area[a]	2.02×10^{10} m^2	
	Roanoke Arm[b]	Nutbush Arm[b]
Water Body Surface Area	1.2×10^8	5.0×10^7 m^2
Mean Depth	10.3	8.2 m
Hydraulic Residence Time	0.2	5.1 yr
Mean Alkalinity[c]	28	22 mg/l as $CaCO_3$
Mean Conductivity[c]	100	123 μmhos/cm @ 25°C
Mean Secchi Depth[c]	1.4	1.2 m
Mean Dissolved Phosphorus[c]	0.01	0.02 mg P/l
Mean Total Phosphorus[c]	0.03	0.03 mg P/l
Mean Inorganic Nitrogen[c] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.28	0.22 mg N/l
Mean Chlorophyll a[c]	13.2	21.2 μg/l
Annual Primary Productivity[c]	171	249 g $C/m^2/yr$
Phosphorus Loading:		
Point Source	630,600	30,500 kg P/yr
Non-Point Source	13,600	5,500 kg P/yr
Surface Area Loading	5.2	0.7 g $P/m^2/yr$
Nitrogen Loading:		
Point Source	18,500	7,480 kg N/yr
Non-Point Source	4,509,600	114,400 kg N/yr
Surface Area Loading	36.2	2.4 g $N/m^2/yr$

	Growing Season Mean Epilimnetic Values		Spring Overturn Mean Values	
	Roanoke Arm	Nutbush Arm	Roanoke Arm	Nutbush Arm
Total Phosphorus (mg P/l)	0.02	0.03	0.04	0.05
Dissolved Phosphorus (mg P/l)	0.006	0.007	0.006	0.010
Inorganic Nitrogen (mg N/l)	0.13	0.10	0.30	0.20
Chlorophyll a (μg/l)	14	18	-	-
Primary Productivity (g C/m^2/day)	0.7	0.7	-	-

Mean Hypolimnetic D.O. Content (mg/l):	3/14/74	5/6/74	7/3/74
Roanoke Arm	9.6	6.8	0.3
Nutbush Arm	10.8	5.1	1.1

(continued)

DATA SUMMARY FOR KERR RESERVOIR (N. CAROLINA-VIR.)* - (continued)

Investigator-Indicated Comments

The upper ends of both arms of the reservoir are nitrogen-limited, while the lower ends of both arms are phosphorus-limited, with respect to algal nutrient requirements.

[a]Does not include water body surface area.

[b]The two principal arms of the impoundment have been treated separately.

[c]Data based on samples obtained at approximately three-month intervals at four stations, six miles apart in the Roanoke Arm, and five stations, three-five miles apart in the Nutbush Arm, during the period 1971-1974. All loading estimates for April, 1974, March, 1975 are based on monthly sampling frequency for all principal phosphorus inputs.

Dash(-) indicates data not available.

*Data taken from Weiss and Moore (1975) and personal communication (Table 3).

LAMB (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Oligotrophic in 1971-1972	
Drainage Area[a]	$2.0 \times 10^6\ m^2$	
Water Body Surface Area	$4.0 \times 10^5\ m^2$	
Mean Depth	4.0 m	
Hydraulic Residence Time	2.3 yr	
	1971	1972
Mean Alkalinity	30	36 mg/l as $CaCO_3$
Mean Conductivity	47	47 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.8	2.2 m
Mean Dissolved Phosphorus	-	-
Mean Total Phosphorus[b]	0.013	0.012 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.51	- mg N/l
Mean Chlorophyll *a*	6 (5)[c]	3 (3)[c] µg/l
Annual Primary Productivity	-	-
Phosphorus Loading:		
Point Source	-	0 kg P/yr
Non-Point Source	-	12.1 kg P/yr
Surface Area Loading	-	$0.03\ g\ P/m^2/yr$
Nitrogen Loading:		
Point Source	-	-
Non-Point Source	-	-
Surface Area Loading	-	-

Investigator-Indicated Comments

Lake highly colored by humic materials. Green and blue-green algae dominates summer and fall phytoplankton community.

[a]Does not include water body surface area.

[b]May-October mean values for 1971-1972.

[c]Euphotic zone.

Dash (-) No data available.

*Data taken from Tarapchak *et al.* (1975) and personal communication (Table 3).

MEANDER (MINN.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Oligotrophic in 1971-1972	
Drainage Area[a]	1.7×10^6 m^2	
Water Body Surface Area	3.6×10^5 m^2	
Mean Depth	5.0 m	
Hydraulic Residence Time	2.7 yr	
	1971	1972
Mean Alkalinity	8	8 mg/l as $CaCO_3$
Mean Conductivity	20.4	16.7 μmhos/cm @ 25°C
Mean Secchi Depth[b]	3.1	3.0 m
Mean Dissolved Phosphorus	-	-
Mean Total Phosphorus[b]	0.012	0.009 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.45	- mg N/l
Mean Chlorophyll *a*[b]	5 (3)[c]	2 (1)[c] μg/l
Annual Primary Productivity	-	-
Phosphorus Loading:		
Point Source	-	0 kg P/yr
Non-Point Source	-	9.9 kg P/yr
Surface Area Loading	-	0.03 g $P/m^2/yr$
Nitrogen Loading:		
Point Source	-	-
Non-Point Source	-	-
Surface Area Loading	-	-

Mean pH = 5.5

Investigator-Indicated Comments

Chrysophytes and crytomonads characterize phytoplankton, except during summer and fall when green and blue-green algae are dominant.

[a]Does not include water body surface area.

[b]May-October mean values.

[c]Euphotic zone.

Dash (-) indicates no data available.

*Data taken from Tarapchak et al. (1975) and personal communication (Table 3).

MENDOTA (WISC.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1965-1966
Drainage Area[a]	6.9×10^8 m^2
Water Body Surface Area	3.9×10^7 m^2
Mean Depth	12 m
Hydraulic Residence Time	4.5 yr
Mean Alkalinity	160 mg/l as $CaCO_3$
Mean Conductivity	300 µmhos/cm @ 25°C
Mean Secchi Depth[b]	3.0 m
Mean Dissolved Phosphorus[b]	0.12 mg P/l
Mean Total Phosphorus[b]	0.15 mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+NO_2^- as N)	0.64 mg N/l
Mean Chlorophyll *a*[c]	10 (20)[d] µg/l
Annual Primary Productivity	1100[e] g C/m^2/yr
Phosphorus Loading:	
Point Source	908[f] kg P/yr
Non-Point Source	45,600 kg P/yr
Surface Area Loading	1.2 g P/m^2/yr
Nitrogen Loading:	
Point Source	3130[f] kg N/yr
Non-Point Source	540,700 kg N/yr
Surface Area Loading	13 g N/m^2/yr

Euphotic zone = to 3 m depth

Euphotic zone volume = 9×10^7 m^3

Summer eiplimnion mean depth = to 10 m

Summer epilimnion mean volume = $3 \times 10^8 \times$ m^3

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Based on 1965-1966 study by students and staff of Water Chemistry Program, Univ. of Wisconsin, Madison, and compiled by Lee (1966).

[c]Mean epilimnetic concentration.

(continued)

DATA SUMMARY FOR MENDOTA (WISC.)* - (continued)

[d]Growing season concentration.

[e]Estimated from chlorophyll and light intensity data.

[f]Point source loadings are mainly storm water drainage inputs.

*Data taken from Lopez and Lee (1975) and personal communication (Table 3).

MICHIGAN (MICH.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Nearshore-Mesotrophic in 1972 Open waters-Oligotrophic in 1974	
Drainage Area[a]	1.8×10^{11} m^2	
Water Body Surface Area	5.8×10^{10} m^2	
Mean Depth	84 m	
Hydraulic Residence Time	30-100 yr	
	Nearshore[b]	Open Waters
Mean Alkalinity	107	113 mg/l as $CaCO_3$
Mean Conductivity	265	255 μmhos/cm @ 25°C
Mean Secchi Depth	2.3	- m
Mean Dissolved Phosphorus	<0.002	0.001[c] mg P/l
Mean Total Phosphorus	0.015	0.013[c] mg P/l
Mean Inorganic Nitrogen ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.20	0.17[c] mg N/l
Mean Chlorophyll *a*	5	2[c] μg/l
Annual Primary Productivity	187-247	150[d] g $C/m^2/yr$
Phosphorus Loading[e]:		
Point Source	-	3.4×10^6 kg P/yr
Non-Point Source	-	2.2×10^6 kg P/yr
Surface Area Loading	- 1971 = 1974 =	0.14 g $P/m^2/yr$ 0.10 g $P/m^2/yr$
Nitrogen Loading[e]:		
Point Source	-	-
Non-Point Source	-	-
Surface Area Loading	- 1971 =	1.3 g $N/m^2/yr$

Euphotic zone = 8 m

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at the four meter depth from one station over an 18 month period in 1970-1971.

[c]After Schelske and Callender (1970).

[d]After Vollenweider (1975a).

[e]After Lee (1974a).

Dash(-) indicates no data available.

*Data taken from Piwoni *et al.* (1976) and personal communication (Table 3).

LOWER LAKE MINNETONKA (MINN.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

	1969	1973
Trophic State	Eutrophic[a] in 1973	
Drainage Area [b]	$3.7 \times 10^8\ m^2$	
Water Body Surface Area	$2.62 \times 10^7\ m^2$	
Mean Depth	8.3 m	
Hydraulic Residence Time	6.3[c] yr	
Mean Alkalinity		- mg/l as $CaCO_3$
Mean Conductivity	125	125 µmhos cm @ 25°C
Mean Secchi Depth [d]	1.5	1.8 m
Mean Dissolved Phosphorus	-	0.003 mg P/l
Mean Total Phosphorus [d]	0.06	0.05 mg P/l
Mean Inorganic Nitrogen ($NH_4^+ + NO_3^- + NO_2^-$ as N)	-	-
Mean Chlorophyll a [d]	21	12 µg/l
Annual Primary Productivity [e]	440	320 g $C/m^2/yr$
Phosphorus Loading:		
Point Source	8900	0 kg P/yr
Non-Point Source	4000	2800 kg P/yr
Surface Area Loading	0.5	0.1 g $P/m^2/yr$ (0.2)[f]
Nitrogen Loading:		
Point Source	-	-
Non-Point Source	-	-
Surface Area Loading	-	-

	Growing Season Mean Epilimnetic Values		Spring Overturn Mean Values	
	1969	1973	1972	1973
Secchi Depth (m)	1.4	1.7	-	-
Total Phosphorus (mg P/l)	0.05	0.04	0.08	0.04
Chlorophyll a (µg/l)	23	15	-	-
Photosynthetic Rate (g C/m^2/day)	2.5	1.9	-	-

(continued)

DATA SUMMARY FOR LOWER LAKE MINNETONKA (MINN.)*- (continued)

Investigator-Indicated Comments

[a]Trophic status as of 1973. Sewage diversion was begun during winter of 1971-1972, eliminating the point source phosphorus input. Prior to sewage diversion, lake was considered eutrophic. Lower Lake Minnetonka is still considered eutrophic in 1973. However, the decreasing nutrient and chlorophyll concentrations and primary productivity and increasing Secchi depth observed in 1973-1974, relative to the 1969 values, indicate the lake to be changing to a less fertile trophic condition.

[b]Does not include water body surface area.

[c]Watershed area and hydraulic residence time data is for entire lake. All other data is only for Lower Lake Minnetonka. It was not possible to calculate hydraulic retention times for individual basins. Thus, the hydraulic residence time for the whole lake was used in all calculations.

[d]Data obtained from samples obtained during the 210-day ice-free period, on ten dates in 1969 and seven dates in 1973, at five meter depth intervals from the surface to the bottom of the lake.

[e]Data obtained from samples incubated at 0, 0.5, 1.0, 2.0, 3.0 and 5.0 meter depths on eight dates between April 25-November 11, 1969 and 1973.

[f]Data in parentheses represents data received by these reviewers from the principal investigator subsequent to completion of this report. Examination of the revised data indicated no significant changes in the overall conclusions concerning Lake Minnetonka.

Dash (-) indicates no data available.

*Data taken from Megard (1975) and personal communication

POTOMAC ESTUARY (MARYLAND, VIRGINIA)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(ESTUARY)

Trophic State	Ultra-eutrophic in 1966-1970		
Drainage Area[a]	3.8×10^{10} m^2		
Water Body Surface Area[b]	Upper Reach	Middle Reach	Lower Reach
	5.7×10^7	2.1×10^8	7.0×10^8 m^2
Mean Depth	4.8	5.1	7.2 m
Hydraulic Residence Time	0.04	0.18	0.85 yr
Mean Alkalinity	70-110	60-85	65-85 mg/l as $CaCO_3$
Mean Conductivity	200-500	600-17,000	17,000-26,000 μmhos/cm@25°C
Mean Secchi Depth[c]	0.4-0.8	0.5-1.3	1.0-2.3 m
Mean Dissolved Phosphorus[c]	0.2-0.8	0.08-0.15	0.01-0.04 mg P/l
Mean Total Phosphorus[c]	0.3-1.2	0.01-0.75	0.03-0.06 mg P/l
Mean Inorganic Nitrogen[c] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	1.8-3.2	0.15-0.33	0.05-0.15 mg N/l
Mean Chlorophyll *a*	30-150	30-100	10-20 μg/l
Annual Primary Productivity	-	-	-
Phosphorus Loading:			
Point Source	4.0×10^6 kg P/yr		
Non-Point Source	8.8×10^5 kg P/yr		
Surface Area Loading	85	8	1.2 g $P/m^2/yr$
	(For total estuary = 5 g $P/m^2/yr$)		
Nitrogen Loading:			
Point Source	9.9×10^6 kg N/yr		
Non-Point Source	6.6×10^6 kg N/yr		
Surface Area Loading	288	32	25 g $N/m^2/yr$
	(For total estuary = 17.2 g $N/m^2/yr$		

Investigator-Indicated Comments

Lower estuary is saline.

Dominant algae is Anacystis.

The dissolved oxygen content is low in the upper and lower reach. The upper and middle reaches become nitrogen-limited with respect to aquatic plant nutrient requirements during the summer months.

[a]Does not include water body surface area.

[b]The estuary has been divided into three separate regions (reaches). Each reach is treated separately.

(continued)

DATA SUMMARY FOR POTOMAC ESTUARY (MARYLAND, VIRGINIA)* - (continued)

[c]June through September values; data based on samples obtained at monthly intervals between 1966-1969, and weekly intervals between 1969-1970, at the top and bottom sampling depths, from sampling stations at five mile intervals in the upper estuary and larger intervals in the lower estuary.

Dash (-) No data available.

*Data taken from Jaworski (1975) and personal communication (Table 3).

LAKE REDSTONE (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	7.7×10^7 m^2
Water Body Surface Area	2.5×10^6 m^2
Mean Depth	4.3 m
Hydraulic Residence Time	0.7-1.0 yr
Mean Alkalinity	125 mg/l as $CaCO_3$
Mean Conductivity	260 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.6 m
Mean Dissolved Phosphorus[b]	0.008[c] mg P/l
Mean Total Phosphorus[b]	0.03[c] mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+NO_2^- as N)	0.80[c] mg N/l
Mean Chlorophyll *a*[b]	12.8 µg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	3630-4230 kg P/yr
Surface Area Loading	1.44-1.68 g P/m^2/yr
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	45,400 kg N/yr
Surface Area Loading	18.1 g N/m^2/yr
Summer Mean Epilimnetic Volumes	
Total Phosphorus	0.11 mg P/l
Dissolved Phosphorus	0.008 mg P/l
Inorganic Nitrogen	0.30 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentration.

Dash (-) No data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

LAKE SALLIE (MINN.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1968-1972
Drainage Area[a]	1.5×10^9 m^2
Water Body Surface Area	5.3×10^6 m^2
Mean Depth	6.4 m
Hydraulic Residence Time	1.1-1.8 yr
Mean Alkalinity	162 mg/l as $CaCO_3$
Mean Conductivity	280-360 μmhos/cm @ 25°C
Mean Secchi Depth	-
Mean Dissolved Phosphorus[b]	0.13 mg P/l
Mean Total Phosphorus[b]	0.35 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.44 mg N/l
Mean Chlorophyll a	-
Annual Primary Productivity	-
Phosphorus Loading[c]:	
Point Source	7060-20,080 kg P/yr
Non-Point Source	1030-1970 kg P/yr
Surface Area Loading	1.5-4.2 g $P/m^2/yr$
Nitrogen Loading[c]:	
Point Source	5590-11,360 kg N/yr
Non-Point Source	4195-9086 kg N/yr
Surface Area Loading	2.8-3.0 g $N/m^2/yr$

	Growing Season (May-September) Mean Epilimnetic Values		Spring Overturn Mean Values
	1972	1973	
Total Phosphorus (mg P/l)	0.4	0.65	1.12
Dissolved Phosphorus (mg P/l)	0.04	0.20	0.26
Inorganic Nitrogen (mg N/l)	0.15	0.18	0.70
Primary Productivity (mg C/m^3/Langley/hr)	9.6	9.6	-

(continued)

DATA SUMMARY FOR LAKE SALLIE (MINN.)* - (continued)

Investigator-Indicated Comments

Hypolimnion does not persist over a growing season

[a]Does not include water body surface area.

[b]Data based on samples obtained at weekly intervals during 1972-1973 at 22 stations located at the lake inlet and outlet, on a transect down the middle of the lake, and around the shore line.

[c]1968-1972 data.

Dash (-) No data available.

*Data taken from Neel (1975) and personal communication (Table 3).

SAMMAMISH (WASH.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Mesotrophic[a] in 1970-1975
Drainage Area[b]	2.7×10^8 m^2
Water Body Surface Area	2.0×10^7 m^2
Mean Depth	18 m
Hydraulic Residence Time	1.8 yr
Mean Alkalinity	33 mg/l as $CaCO_3$
Mean Conductivity	94 μmhos/cm @ 25°C
Mean Secchi Depth	3.3 m
Mean Dissolved Phosphorus[c]	0.006 mg P/l
Mean Total Phosphorus[c]	0.03 mg P/l
Mean Inorganic Nitrogen[c] (NO_3^-+NO_2^- as N)	0.18 mg N/l
Mean Chlorophyll a[c]	5 μg/l
Annual Primary Productivity	238 g C/m^2/yr
Phosphorus Loading:[d]	
Point Source	500 kg P/yr
Non-Point Source	12,500 kg P/yr
Surface Area Loading	0.7 g P/m^2/yr
Nitrogen Loading:[d]	
Point Source	0 kg N/yr
Non-Point Source	258,000 kg N/yr
Surface Area Loading	13.0 g N/m^2/yr

	Growing Season (March - August) Mean Epilimnetic Values	Winter (Dec.-Feb.) Mean Values (photic zone)
Secchi Depth (m)	3.3	3.0
Total Phosphorus (mg P/l)	0.03	0.03
Dissolved Phosphorus (mg P/l)	0.004	0.013
Inorganic Nitrogen (mg N/l)	0.24	-
Chlorophyll a (μg/l)	6	-
Primary Productivity (g C/m^2/day)	0.7	-

Growing Season Hypolimnetic Oxygen Depletion Rate = 0.05 mg/cm^2/day (constant from year to year)

(continued)

DATA SUMMARY FOR SAMMAMISH (WASH.)* - (continued)

Investigator-Indicated Comments

[a]Partial wastewater input diversion (≈30% of total phosphorus input) begun in 1968.

[b]Does not include water body surface area.

[c]Data based on photic zone (7.3 m) measurements.

[d]Post-sewage diversion nutrient loadings. Pre-sewage diversions are as follows: total phosphorus = 20,000 g/yr = 1 $g/m^2/yr$
total nitrogen = 243,000 kg/yr = 12.3 $g/m^2/yr$

Dash (-) No data available.

*Data taken from Welch et al. (1975) and personal communication (Table 3).

SHAGAWA (MINN.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic[a] in 1972
Drainage Area[b]	2.7×10^8 m^2
Water Body Surface Area	9.2×10^6 m^2
Mean Depth	5.7 m
Hydraulic Residence Time	0.8 yr
Mean Alkalinity	22 (fall circulation) mg/l as $CaCO_3$
Mean Conductivity	60 (fall circulation) μmhos/cm @ 25°C
Mean Secchi Depth[c]	2.3 (ice-free period) m
Mean Dissolved Phosphorus[c]	0.021 mg P/l
Mean Total Phosphorus[c]	0.06 mg P/l
Mean Inorganic Nitrogen[c] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.160 mg N/l
Mean Chlorophyll *a*[c]	15 (24)[d] μg/l
Annual Primary Productivity[d]	220 g $C/m^2/yr$
Phosphorus Loading:	
Point Source	5100 kg P/yr
Non-Point Source	1150 kg P/yr
Surface Area Loading	0.7 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	20,000 kg N/yr
Non-Point Source	52,000 kg N/yr
Surface Area Loading	7.8 g $N/m^2/yr$

	Growing Season (May-September) Mean Epilimnetic Values	Spring Overturn Mean Values
Secchi Depth (m)	1.7	2.1
Total Phosphorus (mg P/l)	0.05	0.05
Dissolved Phosphorus (mg P/l)	0.005	0.024
Inorganic Nitrogen (mg N/l)	0.04	0.20
Chlorophyll *a* (μg/l)	31	13.0

1972 Growing Season Hypolimnetic Oxygen Depletion Rate = 1.0 mg/l/week (assumed constant growing season hypolimnion volume)

(continued)

DATA SUMMARY FOR SHAGAWA (MINN.)* - (continued)

Investigator-Indicated Comments

[a]Prior to completion of tertiary waste treatment plant for input wastewater discharges in 1972-1973.

[b]Does not include water body surface area.

[c]Data based on samples obtained from three stations at 1.5 m depth intervals from surface to bottom

[d]Ice-free period averages.

Dash (-) No data available.

*Data taken from Malueg *et al*. (1975) and personal communication (Table 3).

LAKE STEWART (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	2.1×10^6 m^2
Water Body Surface Area	2.5×10^4 m^2
Mean Depth	1.9 m
Hydraulic Residence Time	0.08 yr
Mean Alkalinity	213 mg/l as $CaCO_3$
Mean Conductivity	540 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.4 m
Mean Dissolved Phosphorus[b]	0.001[c] mg P/l
Mean Total Phosphorus[b]	0.04[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	2.26[c] mg N/l
Mean Chlorophyll *a*[b]	12.3 µg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	121-202 kg P/yr
Surface Area Loading	4.82-8.05 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	1850 kg N/yr
Surface Area Loading	73.6 g $N/m^2/yr$
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.08 mg P/l
Dissolved Phosphorus	0.008 mg P/l
Inorganic Nitrogen	0.86 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c]Average winter concentration.

Dash (-) No data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

TAHOE (CALIF., NEVADA)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Ultra-oligotrophic in 1973-1974
Drainage Area[a]	1.3×10^9 m^2
Water Body Surface Area	5.0×10^8 m^2
Mean Depth	313 m
Hydraulic Residence Time	700 yr
Mean Alkalinity	43 mg/l as $CaCO_3$
Mean Conductivity	92 µmhos/cm @ 25°C
Mean Secchi Depth[b]	28.3 m
Mean Dissolved Phosphorus[b]	<0.005 mg P/l (non-detectable)
Mean Total Phosphorus[b]	0.003 mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+NO_2^- as N)	0.02 mg N/l
Mean Chlorophyll a[b]	0.3 µg/l (euphotic zone)
Annual Primary Productivity	5.6 g C/m^2/yr
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	23,400 kg P/yr
Surface Area Loading	0.05 g P/m^2/yr
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	257,300 kg N/yr
Surface Area Loading	0.52 g N/m^2/yr

Growing Season (May-September) Mean Epilimnetic Values:	1973	1974
Secchi Depth (m)	22.5	24.3
Total Phosphorus (mg P/l)	0.003	0.003
Dissolved Phosphorus (mg P/l)	<0.003	<0.003
Inorganic Nitrogen (mg N/l)	0.006	0.003
Chlorophyll a (µg/l)	-	0.2
Primary Productivity (g C/m^2/day)	0.05	0.03
	(6 year euphotic zone average = 0.15)	

(continued)

DATA SUMMARY FOR TAHOE (CALIF., NEVADA)* - (continued)

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at monthly intervals during 1973-1974, at the deep midlake stations from twelve depths between 0 and 400 meters. The chlorophyll value is only for 1974.

[c]Six-year average value

[d]Data based on samples obtained weekly to tri-monthly between August, 1967 and December, 1971 at 13 depths between 0 and 105 m (euphotic zone).

Dash (-) No data available.

*Data taken from Goldman (1975) and personal communication (Table 3).

EAST TWIN LAKE (OHIO)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	[a]Eutrophic in 1972-1974				
Drainage Area[b]	3.3×10^6 m^2				
Water Body Surface Area	2.7×10^5 m^2				
Mean Depth	5.0 m				
	1971	1972	1973	1974[c]	
Hydraulic Residence Time	-	0.8	0.9	0.5	yr
Mean Alkalinity	-	-	105	105	mg/l as $CaCO_3$
Mean Conductivity	-	374	380	366	μmhos/cm @ 25°C
Mean Secchi Depth[d]	2.1	1.6	2.3	1.9	m
Mean Dissolved Phosphorus[d]	0.05	0.03	0.04	0.02	mg P/l
Mean Total Phosphorus[d]	0.09	0.08	0.08	0.08	mg P/l
Mean Inorganic Nitrogen[d] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	1.34	0.58	0.84	-	mg N/l
Mean Chlorophyll a[d]	21	26	22	28	μg/l
Annual Primary Productivity	-	-	-	474[e]	g $C/m^2/yr$
Phosphorus Loading:					
Point Source	-	0	0	0	kg P/yr
Non-Point Source	-	192 (181)[f]	139(127)	185(220)	kg P/yr
Surface Area Loading	-	0.7 (0.7)[f]	0.5(0.5)	0.7(0.8)	g $P/m^2/yr$
Nitrogen Loading:					
Point Source	-	0	0	-	kg N/yr
Non-Point Source	-	8340	5190	-	kg N/yr
Surface Area Loading	-	31.4	19.3	-	g $N/m^2/yr$

Investigator Indicated Comments

[a]Sewage diversion begun in late 1971-1972. Lake was considered early eutrophic prior to sewage diversion. Lake is still considered eutrophic at present time. However, the changing character of the plankton populations indicate the lake to be changing toward a mesotrophic state.

[b]East Twin Lake and West Twin Lake are connected by a tributary and share the same watershed drainage area. Drainage area does not include water body surface area.

[c]Experienced sewage leak from West Twin Lake into East Twin Lake in 1974.

(continued)

[d]Data based on samples obtained from the deepest point in each lake, generally weekly from late spring - early fall, and less frequently the rest of the year, at 0.1, 2, 4, 7, and 10 meters from 1971-1974.

[e]Average of 6 measurements made between June 27, 1974 - August 9, 1974. An *in situ* measurement technique used because of difficulty of estimating primary productivity of extensive macrophyte production.

Summer season mean epilimnetic nutrient concentrations given in Cooke *et al*. (1975)

Dash (-) No data available.

[f]All data in parentheses represents data received by these reviewers from the principal investigators subsequent to completion of this report. The original data supplied by the investigator was used in all figures in this report. Examination of the revised data indicated no significant changes in the overall conclusions concerning East Twin Lake.

*Data taken from Cooke *et al*. (1975) and personal communication (Table 3).

WEST TWIN LAKE (OHIO)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic[a] in 1974				
Drainage Area[b]	3.3×10^6 m^2				
Water Body Surface Area	3.4×10^5 m^2				
Mean Depth	4.34 m				
	1971	1972	1973	1974	
Hydraulic Residence Time	-	1.6	1.8	1.0	yr
Mean Alkalinity	-	-	110	106	mg/l as $CaCO_3$
Mean Conductivity	-	411	409	380	μmhos/cm @ 25°C
Mean Secchi Depth[c]	1.7	2.2	2.8	2.3	m
Mean Dissolved Phosphorus[c]	0.07	0.06	0.06	0.04	mg P/l
Mean Total Phosphorus[c]	0.15	0.12	0.11	0.10	mg P/l
Mean Inorganic Nitrogen[c] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	1.93	0.79	0.83	-	mg N/l
Mean Chlorophyll *a*[c]	27	40	23	28	μg/l
Annual Primary Productivity	-	-	-	576[d]	g C/m^2/yr
Phosphorus Loading:					
Point Source	-	0	0	0	kg P/yr
Non-Point Source	-	118(143)[d]	103(61)	91(107)	kg P/yr
Surface Area Loading	-	0.4(0.4)	0.3(0.2)	0.3(0.3)	g P/m^2/yr
Nitrogen Loading:					
Point Source	-	0	0	-	
Non-Point Source	-	5457	5094	-	kg N/yr
Surface Area Loading	-	16	15	-	g N/m^2/yr

Investigator-Indicated Comments

[a]Sewage diversion begun in late 1971-1972. Lake was considered eutrophic prior to sewage diversion. However, lake is considered mesotrophic at the present time because of its changing plankton characteristics.

[b]East Twin and West Twin Lake are connected by a tributary and share the same watershed drainage area. Drainage area does not include water body surface area.

[c]Data based on samples obtained from the deepest point in each lake, generally weekly from late spring-early fall, and less frequently the rest of the year at 0.1, 2, 4, 7 and 10 meters from 1971-1974.

(continued)

[d]All data in parentheses represents data received by these reviewers from the principal investigator subsequent to completion of this report. The original data supplied by the investigator was used in all figures in this report. Examination of the revised data indicated no significant changes in the overall conclusions concerning West Twin Lake.

Summer season mean epilimnetic nutrient concentrations given in Cooke et al. (1975).

Dash (-) No data available.

*Data taken from Cooke et al. (1975) and personal communication (Table 3).

TWIN VALLEY LAKE (WISC.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	3.1×10^7 m^2
Water Body Surface Area	6.1×10^5 m^2
Mean Depth	3.8 m
Hydraulic Residence Time	0.4-0.5 yr
Mean Alkalinity	175 mg/l as $CaCO_3$
Mean Conductivity	370 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.5 m
Mean Dissolved Phosphorus[b]	0.019[c] mg P/l
Mean Total Phosphorus[b]	0.07[c] mg P/l
Mean Inorganic Nitrogen[b] (NH_4^++NO_3^-+NO_2^- as N)	0.51[c] mg P/l
Mean Chlorophyll a[b]	19 µg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	1090-1250 kg P/yr
Surface Area Loading	1.74-2.05 g P/m^2/yr
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	10,500 kg N/yr
Surface Area Loading	17.4 g N/m^2/yr
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.06 mg P/l
Dissolved Phosphorus	0.01 mg P/l
Inorganic Nitrogen	0.23 mg N/l

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at six week intervals, at either one or two meter depth intervals, in the deepest part of the impoundment.

[c]Average winter concentrations.

Dash (-) No data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

LAKE VIRGINIA (WISC.)*

DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT

(SEEPAGE IMPOUNDMENT)

Trophic State	Eutrophic in 1972-1973
Drainage Area[a]	6.5×10^6 m^2
Water Body Surface Area	1.8×10^5 m^2
Mean Depth	1.7 m
Hydraulic Residence Time	0.9-2.8 yr
Mean Alkalinity	64 mg/l as $CaCO_3$
Mean Conductivity	230 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.2 m
Mean Dissolved Phosphorus[b]	0.004[c] mg P/l
Mean Total Phosphorus[b]	0.02[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.22[c] mg P/l
Mean Chlorophyll a[b]	29.0 µg/l (first two meters of water column)
Annual Primary Productivity	-
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	210-270 kg P/yr
Surface Area Loading	1.15-1.48 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	3300 kg N/yr
Surface Area Loading	18.3 g $N/m^2/yr$
Summer Mean Epilimnetic Values:	
Total Phosphorus	0.15 mg P/l
Dissolved Phosphorus	0.025 mg P/l
Inorganic Nitrogen	0.18 mg N/l

Investigator-Indicated Comments

[a] Does not include water body surface area.

[b] Data based on samples obtained at six week intervals at either one or two meter depth intervals in the deepest part of the impoundment.

[c] Average winter concentration.

Dash (-) No data available.

*Data taken from Piwoni and Lee (1975) and personal communication (Table 3).

WALDO (ORE.)*
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Ultra-Oligotrophic in 1974
Drainage Area[a]	7.9×10^7 m^2
Water Body Surface Area	2.7×10^7 m^2
Mean Depth	36 m
Hydraulic Residence Time	21 yr
Mean Alkalinity	1.8 mg/l as $CaCO_3$
Mean Conductivity	3.4 μmhos/cm @ 25°C
Mean Secchi Depth[b]	28 m
Mean Dissolved Phosphorus[b]	<0.005 mg P/l
Mean Total Phosphorus[b]	<0.005 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	<0.010 mg N/l
Mean Chlorophyll <u>a</u>	0.32[c] μg/l
Annual Primary Productivity[e]	0.001-0.003 g $C/m^2/day$[d]
Phosphorus Loading[e] :	
Point Source	0 kg P/yr
Non-Point Source	458 kg P/yr
Surface Area Loading	0.017 g $P/m^2/yr$
Nitrogen Loading[f]:	
Point Source	0 kg N/yr
Non-Point Source	9020 kg N/yr
Surface Area Loading	0.33 g $N/m^2/yr$

<u>Investigator-Indicated Comments</u>

[a]Does not include water body surface area.

[b]Data based on samples obtained from nine stations each August from 1970 to 1974, at 20 meter depth intervals. Significant differences between epilimnetic and hypolimnetic values do not appear to exist.

[c]Average of summer measurements for 1969, 1970, and 1974.

[d]Summer 1970 value.

[e]Based on average of four indirect calculation methods. (see Powers <u>et al</u>., 1975)

[f]Based on average of two indirect calculation methods. (see Powers <u>et al</u>., 1975)

*Data taken from Powers <u>et al</u>. (1975) and personal communication (Table 3).

WASHINGTON (WASH.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Mesotrophic[a] in 1974
Drainage Area[b]	1.6×10^9 m^2
Water Body Surface Area	8.8×10^7 m^2
Mean Depth	33 m
Hydraulic Residence Time	2.4 yr
Mean Alkalinity	45 mg/l as $CaCO_3$
Mean Conductivity	81 μmhos/cm @ 25°C

	1933	1957	1963-4	1971	1974	
Mean Secchi Depth	-	2.2	1.2	3.5	3.8	m
Mean Dissolved Phosphorus	0.003	0.002	0.030	0.006	(3.5 m)[e]	
Mean Total Phosphorus	0.016	0.024	0.066	0.018	-	mg P/l
Mean Inorganic Nitrogen ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.007	0.12	0.24	0.18	-	mg N/l
Mean Chlorophyll *a*	-	12	20	6	(4)[e]	
Annual Primary Productivity	-	-	766	354	-	g C/m^2/yr

	1957[c]	1964[c]	1971[d]	1974[d]	
Phosphorus Loading:					
Point Source	57,100	103,900	0	0	kg P/yr
Non-Point Source	60,400	98,500	37,600	41,300	kg P/yr
Surface Area Loading	1.2	2.3	0.43	0.47	g P/m^2/yr
Nitrogen Loading:					
Point Source	201,700	271,000	0	0	kg N/yr
Non-Point Source	1,487,200	418,200	401,600	386,900	kg N/yr
Surface Area Loading	19.2	7.8	4.6	4.4	g N/m^2/yr

Growing Season (May-Sept.)

Mean Epilimnetic Values:	1933	1957	1963	1971
Total Phosphorus (mg P/l)	0.013	0.022	0.060	0.014
Dissolved Phosphorus (mg P/l)	0.001	0.002	0.010	0.005
Inorganic Nitrogen (mg N/l)	0.037	0.042	0.106	0.067
Chlorophyll *a* (μg/l)	-	15	29	9

(continued)

DATA SUMMARY FOR WASHINGTON (WASH.) * - (continued)

Investigator-Indicated Comments

[a]Sewage diversion project begun in 1963 and completed in 1968. Lake Washington was considered eutrophic prior to 1963. However, the nutrient and chlorophyll concentrations and primary productivity have decreased dramatically since 1963, indicating a much lower fertility as a result of the sewage diversion project. Lake Washington is considered mesotrophic at the present time.

[b]Does not include water body surface area.

[c]Maximum estimated input, including septic tank drainage. However, part of this would already have been measured in the stream inputs, and therefore this estimate may be slightly higher than the actual input phosphorus loading.

[d]Post-sewage diversion loading of the two major outlets; does not include storm water drainage overflow, which is not considered a major nutrient input source.

[e]Data in parentheses represents data received by these reviewers from the principal investigators subsequent to completion of this report. Examination of the data indicates no significant changes in the overall conclusions concerning the water body.

Dash (-) No data available.

*Data taken from Edmondson (1975a) and personal communication (Table 3).

WEIR (FLA.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Mesotrophic in 1974-75
Drainage Area[a]	$4.6 \times 10^7\ m^2$
Water Body Surface Area	$2.4 \times 10^7\ m^2$
Mean Depth	6.3 m
Hydraulic Residence Time	4.2 yr
Mean Alkalinity[b]	11.5 mg/l as $CaCO_3$
Mean Conductivity[b]	133 µmhos/cm @ 25°C
Mean Secchi Depth[b]	1.9 m
Mean Dissolved Phosphorus[b]	0.025 (0.006)[c] mg P/l
Mean Total Phosphorus[b]	0.08 (0.02)[c] mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^- + NO_2^-$ as N)	0.07 (0.20)[c] mg N/l
Mean Chlorophyll *a*	8 (6)[c] µg/l
Annual Primary Productivity	36 g $C/m^2/yr$
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	3290 kg P/yr
Surface Area Loading	0.14 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	61,920 kg N/yr
Surface Area Loading	2.6 g $N/m^2/yr$
May-September Mean Epilimnetic Values:	
Secchi Depth	1.9 m
Total Phosphorus	0.08 mg P/l
Dissolved Phosphorus	0.022 mg P/l
Inorganic Nitrogen	0.04 mg N/l
Chlorophyll *a*	4 µg/l
Primary Productivity	0.4 g $C/m^2/day$

(continued)

DATA SUMMARY FOR WEIR (FLA.)* - (continued)

Investigator-Indicated Comments

[a]Does not include water body surface area.

[b]Data based on samples obtained at biweekly intervals at three stations at the surface, 1m, 3m, 5m, and at station 1, 7m depths, from 6/20/74 to 1/19/75.

[c]1969-70 average values.

*Data taken from Brezonik and Messer (1975) and personal communication (Table 3).

WINGRA (WISC.)
DATA SUMMARY FOR US OECD EUTROPHICATION PROJECT
(LAKE)

Trophic State	Eutrophic in 1970-1971
Drainage Area[a]	$1.4 \times 10^7\ m^2$
Water Body Surface Area	$1.4 \times 10^6\ m^2$
Mean Depth	2.4 m
Hydraulic Residence Time	0.4 yr
Mean Alkalinity	153 mg/l as $CaCO_3$
Mean Conductivity	-
Mean Secchi Depth[b]	1.3 m
Mean Dissolved Phosphorus[b]	0.02 mg P/l
Mean Total Phosphorus[b]	0.07 mg P/l
Mean Inorganic Nitrogen[b] ($NH_4^+ + NO_3^-$ as N)	0.31 mg N/l
Mean Chlorophyll a	-
Annual Primary Productivity	870 g $C/m^2/yr$ (phytoplankton productivity)
Phosphorus Loading:	
Point Source	0 kg P/yr
Non-Point Source	1200 kg P/yr
Surface Area Loading	0.9 g $P/m^2/yr$
Nitrogen Loading:	
Point Source	0 kg N/yr
Non-Point Source	7200 kg N/yr
Surface Area Loading	5.14 g $N/m^2/yr$
Growing Season (May-September)	
Mean Epilimnetic Values:	
Total Phosphorus	0.08 mg P/l
Dissolved Phosphorus	0.06 mg P/l
Inorganic Nitrogen	1.0 mg N/l
Primary Productivity	4.6 g $C/m^2/day$

Investigator-Indicated Comments

Lake has extensive littoral zone and exhibits large amount of macrophyte growth.

(continued)

DATA SUMMARY FOR WINGRA (WISC.)* - (continued)

[a]Does not include water body surface area.

[b]Data based on samples obtained at weekly intervals during 1970-1971, at one and two meters, from four open lake and four littoral zone stations.

Dash (-) No data available.

*Data taken from Rast and Lee (1975) and personal communication (Table 3).

GLOSSARY

A_d	Watershed area (L^2)
A_o	Water body surface area (L^2)
L	Areal loading ($ML^{-2}T^{-1}$)
L_c	Critical loading ($ML^{-2}T^{-1}$)
$L_c(P)$	Permissible ("critical") total phosphorus loading ($ML^{-2}T^{-1}$)
$L(N)$	Total nitrogen loading ($ML^{-2}T^{-1}$)
$L(P)$	Total phosphorus loading ($ML^{-2}T^{-1}$)
$L(P)/q_s$	Influent total phosphorus concentration (ML^{-3})
$\ell(P)$	Volumnar total phosphorus loading ($ML^{-3}T^{-1}$) = $L(P)/\bar{z}$
MDR	Meteoric discharge rate (LT^{-1})
$[P]$ & $[P]\lambda$	In-lake total phosphorus concentration (ML^{-3})
$[P]$	Influent total phosphorus concentration (ML^{-3}) = $L(P)/q_s$
$[P]_j$	Inflow total phosphorus concentration (ML^{-3})
$[P]_o$	Outflow total phosphorus concentration (ML^{-3})
$[P]_c^{sp}$	Critical total phosphorus concentration at spring overturn (ML^{-3})
$[P]_\lambda^{summer}$	Summer mean in-lake total phosphorus concentration (ML^{-3})
$[P]_t$	Total phosphorus concentration at time t (ML^{-3})
$[P]_{t_o}$	Total phosphorus concentration at time 0 (ML^{-3})
$[P]_\infty$	Steady state total phosphorus concentration (ML^{-3})

Q	Annual inflow or outflow volume (L^3)
q_i	Inflow volume (L^3)
q_o	Outflow volume (L^3)
q_s	Hydraulic loading (LT^{-1}) = $\bar{z}/\tau_\omega$ = areal water load (Q/A_o)
R	Retention coefficient
R_n	Nitrogen residence time (T)
R_p	Phosphorus residence time (T)
$1-R(P)$	Fraction of phosphorus input not retained in sediment
$\bar{\tau}_p$	Phosphorus residence time (T)
V	Water body volume (L^3)
ν	Apparent settling velocity of total phosphorus (LT^{-1}) = $\propto \nu^1$
ν_j	Flow rate in j^{th} tributary (L^3T^{-1})
ν^1	Settling velocity of settleable particulate phosphorus (LT^{-1})
$\bar{z}$	Mean depth (L) = V/A_o
$\propto$	Fraction of total phosphorus represented by settleable particulate phosphorus
ρ	Flushing rate (T^{-1})
ρ_ω	Hydraulic flushing rate (T^{-1})
π_r	Phosphorus residence time relative to hydraulic residence time (TT^{-1}) = $\bar{\tau}_p/\tau_\omega$
σ	Sedimentation rate coefficient (T^{-1})
σ_p	Phosphorus sedimentation rate coefficient (T^{-1})
τ_ω	Hydraulic residence time (T)

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16. ABSTRACT

This report summarizes and critically analyses nutrient load-lake response relationships for 38 U.S. water bodies which have been intensively studied by 20 scientists participating in the OECD Eutrophication Program.

It was determined that the Vollenweider nutrient load relationship involving mean depth, hydraulic residence time and phosphorus load correlated well with the trophic states assigned by the individual investigators.

A good correlation was also found between phosphorus loading, normalized as to hydraulic residence time and mean depth, and the average chlorophyll and water clarity (as neasured by Secchi depth).

The relationships developed in this study can be used to predict the improvement in water quality that will result from a change in the phospohrus load to a water body for which phosphorus is the key chemical element limiting planktonic algal growth.

17. KEY WORDS AND DOCUMENT ANALYSIS		
a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
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