**REVIEW**

# RECENT ADVANCES IN ASSESSING IMPACT OF PHOSPHORUS LOADS ON EUTROPHICATION-RELATED WATER QUALITY

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## INTRODUCTION

Eutrophication—excessive fertilization, which is manifested by excessive growths of planktonic (suspended) and attached algae, and aquatic macrophytes (water weeds), can have significant deleterious effects on the beneficial uses of lakes, impoundments, estuarine, and marine waters. Lee (1973), who reviewed in detail the causes, impacts and common control options for eutrophication, pointed out that excessive growths of aquatic plants can interfere with the use of waters for domestic and industrial water supply, irrigation, recreation, fisheries, etc. Further. it has recently been determined for some waterbodies that there is an apparent relationship between the degree of eutrophication and the amount of trihalomethanes formed during chlorination of the water during treatment for domestic use. Trihalomethanes are chloroform-like compounds which, if ingested in large amounts, are known to be carcinogenic to animals. The widespread applicability of the apparent relationship between trophic status and trihalomethane formation, however, is not known at this time.

As discussed by Krenkel *et al.* (1979), excessive growths of aquatic plants in a waterbody can also cause deterioration of water quality downstream of the affected waterbody. Of particular concern in this regard is the release from a reservoir of deoxygenated hypolimnetic (bottom) waters which can result from excessive growths of algae. Such releases can cause obnoxious odors as well as impair the fishery potential of downstream waters.

Because of the potential and realized water quality deterioration associated with excessive growths of algae and other aquatic plants, considerable effort is being expended in the U.S. and abroad to control eutrophication. Presented below is a discussion of the recent advances in assessing eutrophication-related water quality and in developing and evaluating eutrophication control options with emphasis on phosphorus load control.

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## APPROACH TO EUTROPHICATION MANAGEMENT

### *Course of Eutrophication*

The most cost-effective approach toward eutrophication management is usually to eliminate or reduce the cause of the excessive aquatic plant growths. Algae, as well as other aquatic plants, need a wide variety of chemical constituents, in addition to sunlight, for growth. Ordinarily, all constituents needed for their optimum growth based on the light available, are present in aquatic systems in surplus amounts compared to the organisms' needs, except for nitrogen and/or phosphorus. It is generally the rate of supply of algal available forms of nitrogen or phosphorus which limits the maximum algal biomass reached in a waterbody. When the load of the nutrient which is limiting algal growth is decreased, a decrease in the maximum algal biomass that is produced would be expected. As discussed herein, it usually requires a fairly substantial reduction in the available load of the limiting nutrient to cause a noticeable improvement in eutrophication-related water quality characteristics. Common sources of aquatic plant nutrients to waterbodies include domestic wastewater treatment plant effluent, direct and Indirect (tributary streams) runoff from land, and atmospheric precipitation and dry fallout. The amounts of N and P that are derived from these sources can be readily and fairly reliably estimated using techniques described in a subsequent section of this paper.

There are several elements which are basic to developing an effective, in terms of both cost and water quality improvement, eutrophication-nutrient management program, each one of which must be evaluated on a site specific basis. First, efforts should be oriented to controlling the nutrient which limits maximum algal biomass in the waterbody of concern. Most commonly, focus is placed on phosphorus control since it most often is the limiting nutrient and methods to control its loading from domestic wastewaters, often a substantial source, are readily available and relatively inexpensive to implement. It is important to note that even in waterbodies in which algal growth is limited by reduction can result in improved water quality if the phosphorus load reduction is sufficiently large to drive the waterbody to P limitation, and then reduced sufficiently beyond that to decrease algal growth. Second, if phosphorus control efforts are practiced, they should be directed toward those sources of phosphorus which permit the greatest reduction in algal available phosphorus and which are most readily controllable. Lee *et al.* (1980) provide a state-of-the-art summary of the availability to stimulate algal growth, of various forms of phosphorus and of the phosphorus derived from various sources, as well as methods of evaluating the availability of phosphorus from a particular source. Finally, a quantitative assessment should be made of the improvement in water quality—beneficial uses of the water that can be achieved by implementing P management options. While some espouse the approach to P load reductions to waterbodies that "every little bit helps", there is no technical justification for that approach and following it will not lead to the most cost-effective, technically sound, yet environmentally protective eutrophication control programs. Rather, it can readily lead to the public's spending large amounts of money in the name of "pollution control" with little improvement in water quality beyond that which could be attained using a rational, technically defensible approach to nutrient load assessment and control. To assess the impact on water quality that will result from a given control program, a verified method should be used to relate the phosphorus load and projected change in load to the eutrophication-related water quality response (in terms of-beneficial uses and public perception) of the waterbody.

Numerous attempts have been and are currently being made to quantify the cause-effect coupling between nutrient load and eutrophication-related water quality response, to create models which could be used as a basis for eutrophication-related water quality management. Such models, if properly formulated and verified, would allow determinations to be made of the changes in water quality that would result from given changes in nutrient load as well as projections to be made of the water quality that will exist in an impoundment that has not yet been created. Most of this modeling effort has been directed towards formulating "dynamic" models which are mathematical representations of the key physical, chemical, and biological processes governing algal growth in waterbodies. While it is possible to develop a set of differential equations which can describe algal population dynamics in relation to nutrient loads for a given set of circumstances; few of these models have demonstrated capability for predicting response when the nutrient loads are substantially altered or when applied to a waterbody other than the one upon which it was developed. The Canada Centre for Inland Waters (1979) has recently conducted a review of "dynamic" eutrophication models developed for Lake Ontario and concluded that such models water quality management purposes. An alternate approach to the dynamic model is the statistical modeling approach such as the Vollenweider-OECD models described below. Occasionally reference will be made to the statistical eutrophication models of Dillon & Rigler (1974) and Larsen & Mercier (1976). As discussed by Rast & Lee (1978), these models have the same technical foundation as the Vollenweider-OECD models. However, the correlations or lines of best fit for these models were not developed on as broad a data base as those of the OECD load-response models discussed in this paper.

### *OECD Eutrophication Modeling Approach*

*Background*

The Organization for Economic Cooperation and Development (OECD) Eutrophication Study was undertaken to quantitatively define the relationship between the nutrient (phosphorus) load to a waterbody (lake, impoundment, or estuary) and the eutrophication-related water quality response of the waterbody to that load. This recently-completed five year study involved the examination of P load and response characteristics of about 200 waterbodies in 22 countries in Western Europe. North America, Japan, and Australia. The 34 U.S. waterbodies or parts thereof included in the OECD Eutrophication Study Program were selected because a considerable body of information on their nutrient loading, hydrological and morphological characteristics was already available and individuals who had investigated these waterbodies were willing to assemble the information on them. G. F. Lee had the contract with the U.S. EPA to critically review these data and evaluate the waterbodies' overall load-response characteristics.

Vollenweider (1975), through his work with the OECD, developed a model which described a relationship between the phosphorus load to a waterbody and the relative general acceptability of the water for recreational use. Figure 1 shows this model applied to the U.S. OECD waterbodies. Rast & Lee (1978) and Lee et al (1978a) found, as Vollenweider (1975) had for a smaller group of primarily European lakes, that when the annual areal P load to a lake or impoundment is plotted as a function of the quotient of the mean depth (volume ÷ surface area) and hydraulic residence time (filling time), waterbodies which were eutrophic (highly fertilized) tended to cluster in one area, as the oligotrophic waterbodies (poorly fertilized—lower amounts of aquatic plant growth) clustered in another area. As the phosphorus load to a P-limited waterbody having a given mean depth/hydraulic residence time quotient is reduced, eutrophication-related water quality is generally improved, provided the reduction is sufficiently large. It is important to note that the “Excessive" and "Permissible" lines in Fig. 1 do not represent sharp

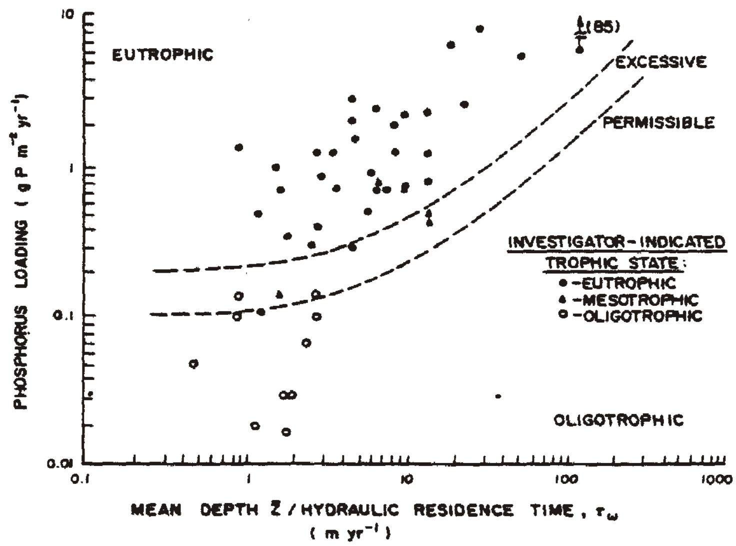



**Fig. 1.** U.S. OECD data applied to Vollenweider P loading–mean depth/hydraulic residence time relationship (after Rast & Lee, 1978).

boundaries of water quality between oligotrophic and eutrophic waters. For any set of waterbodies having a given mean depth/hydraulic residence time quotient, there is a gradation in water quality along the vertical, with waterbodies having better water quality plotting toward the bottom and those having poorer water quality, toward the top.

Vollenweider (1976) subsequently developed a statistical correlation between the areal annual P loading [L(P)] to a waterbody normalized by mean depth (z) and hydraulic residence time ($\tau_\omega$), (i.e. $[L(P)/(z/\tau_\omega)]/[1 + (\tau_\omega)^{1/2}]$), and the eutrophication response of the waterbody as measured by mean chlorophyll concentration, for a small group of waterbodies. Based on the data from the U.S. OECD waterbodies, Rast & Lee (1978) and Lee *et al.* (1978a) substantiated that general relationship and defined it for a greater number of waterbodies, deriving a somewhat different line of best fit than Vollenweider's. They also expanded Vollenweider's concept and developed correlations between the normalized P load and Secchi depth (water clarity), and between the normalized P load and hypolimnetic oxygen depletion rate. Figure 2 shows these three load-response relationships for the U.S. OECD waterbodies.

*Results of recent work*

Since the completion of the U.S. part of the OECD Eutrophication Study, the authors and their associates have evaluated the P load-response relationship for another approx. 40 U.S. waterbodies, including a dozen Tennessee River System (TRS) impoundments, and a variety of waterbodies in Indiana, Wisconsin, Colorado, Wyoming, and Texas. These additional waterbodies were not selected *per se* in any way, but rather are the group of waterbodies which the authors have been able to investigate and evaluate over the past several years for nutrient load-eutrophication response relationships. The results of these specific studies are in various stages of publication and include Lee *et al.* (1978b, 1979, 1980), Newbry *et al.* (1979, 1980), Lee & Meckel (1978), Horstman *et al.* (1980). Lee & Jones (1981c), Jones & Lee (1978), and Lee & Archibald (1981).

The locations of the approx. 30 US. waterbodies evaluated to date are shown in Fig. 3. Figure 4 shows the lines of best fit for the P load-chlorophyll, P load-Secchi depth, and P load-hypolimnetic oxygen depletion rate relationships for these U.S. waterbodies and the associated 95% confidence intervals for the data. (Except for the confidence limits, Figs 2(C) and 4(C) are identical, as no additional P load-hypolimnetic oxygen depletion rate couplings were available.) As can be seen by comparison of Figs 2 and 4, the regressions developed based only on the U.S. OECD data are essentially identical to those developed by including the additional lakes and impoundments.

Further, the remainder of the approx. 200 waterbodies have also been found to follow the general relationships shown in Figs 2 and 4 (Clasen, 1979, Vollenweider & Kerekes, 1980). It therefore appears that these relationships have a widespread applicability to most waterbodies—those in northern as well as southern climates, shallow as well as deep waterbodies, lakes as well as impoundments, large waterbodies and small waterbodies. Contrary to some assertions which have been made by the US. Army Corps of Engineers (1979) for example, and as shown in Fig. 4, the OECD eutrophication studies have demonstrated that as a group. lakes do not respond differently to nutrient input than impoundments: the same load-response regressions are applicable to both lakes and impoundments. However, as discussed below, there is a number of characteristics of impoundments as well as some lakes and estuaries, which may necessitate modification of input parameter values.

Based on the results of the OECD eutrophication study and the subjective opinions of the individual investigators regarding the limnological trophic conditions of their waterbodies, Vollenweider (Vollenweider & Kerekes, 1980) developed distributions of the probability that a waterbody will be classified in a certain limnological trophic category given a certain chlorophyll concentration, Secchi depth, or total P concentration, among other parameters. Following the probability distribution for chlorophyll and through the U.S. waterbody load-response couplings shown in Fig. 4, the authors developed the response parameter values in Table 1 which tend to be indicative of certain trophic designations for waterbodies. It is important to understand however, that these values should not be strictly applied for the classification of waterbodies,

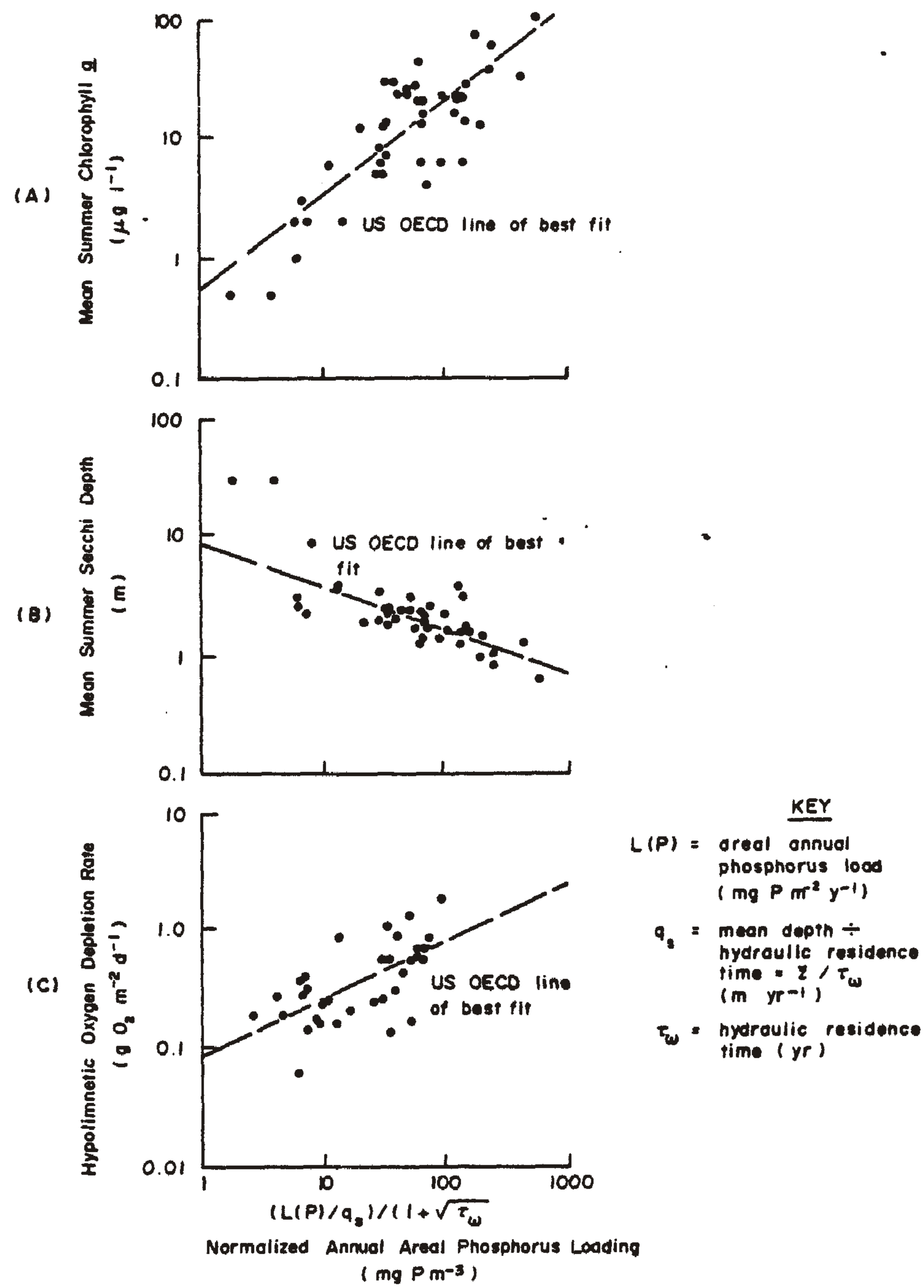



Fig. 2. Normalized P load, chlorophyll concentration, Secchi depth, hypolimnetic oxygen depletion rate relationships based on U.S. OECD data (after Lee *et al.*, 1978a).

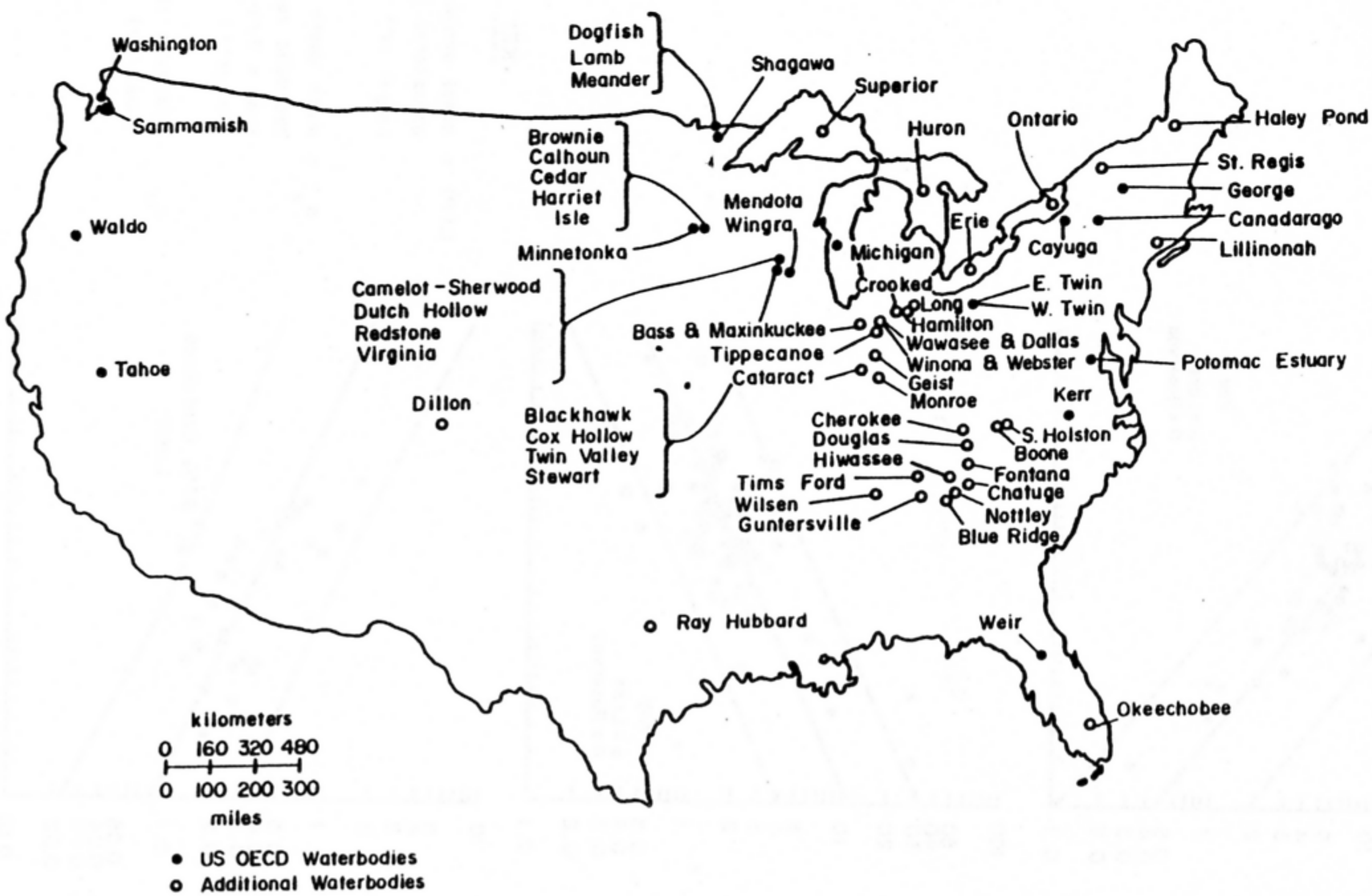



Fig. 3. Locations of U.S. waterbodies for which load-response relationships have been evaluated.

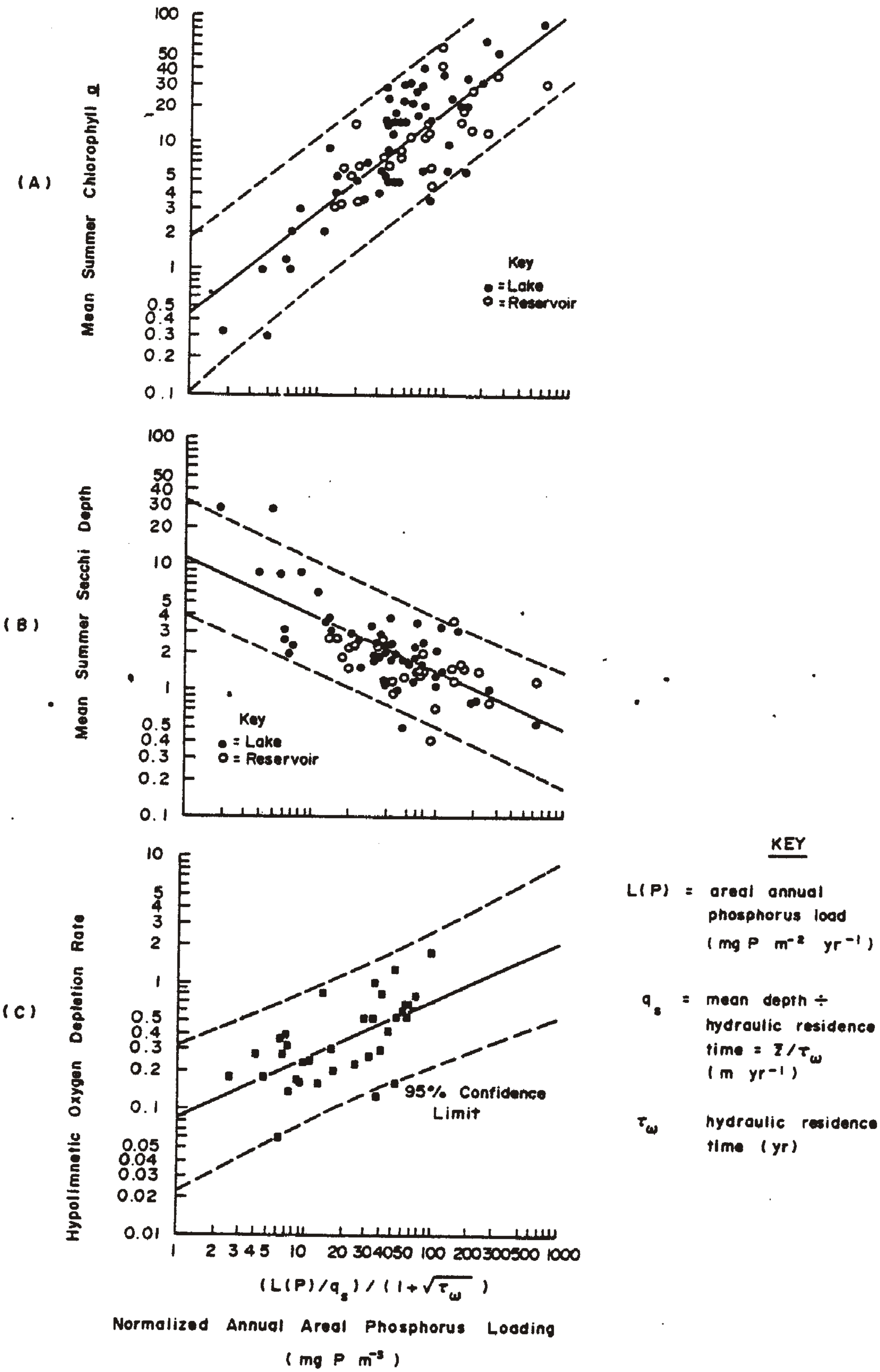



Fig. 4. Updated P load–eutrophication related water quality response relationships for U.S. waterbodies.

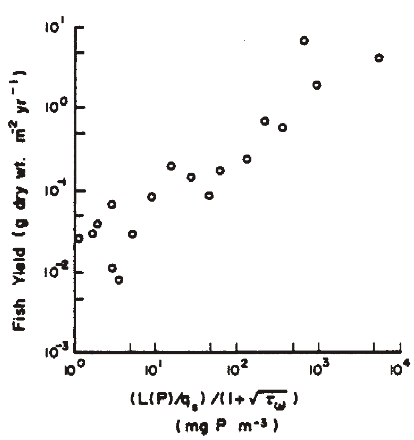
**Table 1. Limnological classification of trophic status of lakes and reservoirs**

| Classification | Average planktonic algal chlorophyll ($\mu g\ l^{-1}$) | Average Secchi depth (m) | Average in lake total phosphorus ($\mu g\ l^{-1}$) |
|---|---|---|---|
| Oligotrophic | $<2.0$ | $>4.6$ | $<7.9$ |
| Oligotrophic-mesotrophic | 2.1–2.9 | 4.5–3.8 | 8–11 |
| Mesotrophic | 3.0–6.9 | 3.7–2.4 | 12–27 |
| Mesotrophic-eutrophic | 7.0–9.9 | 2.3–1.8 | 28–39 |
| Eutrophic | $\geqslant 10$ | $\leqslant 1.7$ | $\geqslant 40$ |

After Lee *et al.* (1981c).

especially for water quality management purposes. Lee *et al.* (1981c) discuss how this table and the OECD eutrophication modeling approach should be used in conjunction with desired beneficial waterbody uses, for water quality management—PL 92-500 Section 314 A classification purposes. They stress the fact that for most applications, planktonic algal chlorophyll concentration tends to be the most reliable eutrophication-related water quality indicator; for those waterbodies having average amounts of inorganic turbidity and color, Secchi depth is an adequate secondary parameter. The total P concentrations in a waterbody should not, in general, be used as an indication of water quality response since, as discussed by Lee *et al.* (1981c) as well as others, unless the P is used in the production of undesirable amounts of algae it does not typically cause an impairment of beneficial uses of the water.

Lee & Jones (1979) have further extended the Vollenweider-Rast & Lee-OECD nutrient




Line of best fit:

$$\text{Log Fish Yield} = 0.7 \log[(L(P)/q_s)/(1+\sqrt{\tau_\omega})] - 1.86$$
$$(r^2 = 0.86)$$

Fig. 5. Relationship between normalized P load and fish yield (after Lee and Jones, 1979).

between P load and fish yield (Fig. 5). While additional data are needed to better define the correlation, the relationship is clear. It should be noted that this relationship does not explicitly address the types or quality of fish present. While the overall fish yield will increase with increased fertilization, there is a point at which in certain waterbodies the aquatic plant growth would be sufficient to significantly deplete hypolimnetic oxygen and preclude the survival of cold water fish types such as salmonids (trout). Further, some highly eutrophic waterbodies tend to have large populations of stunted pan fish. Therefore, a plot of yield of desirable fish vs normalized P loading would likely have two breaks in it, one corresponding to the toad which causes sufficient algal growth to cause complete hypolimnetic oxygen depletion, and the other corresponding to the load resulting in stunted fish growth. It is important to note that while in more developed countries, the management emphasis is on control of eutrophication-related water quality to reduce the production of aquatic plants in order to improve the recreational-aesthetic quality and treatability of the water for drinking, in many lesser developed countries, the emphasis will likely be on maximizing production of fish for food without rendering the waters unsuitable for other purposes. The latter philosophy can now be handled at least in a rudimentary way, through the Lee and Jones relationship (Fig. 5). As discussed above and in detail by Lee & Jones (1979) however, the use of this approach for estimating edible fish abundance must be made cautiously.

Recently, the predictive capability of the OECD modeling approach has been verified by Rast *et* al. (1981). They examined the P loads and waterbody response (chlorophyll) for the approx. 10 waterbodies to which the P loads had been substantially changed, and on which sufficient "before" and "after" P load change data were available. Using the OECD eutrophication modeling approach and pre-change data, the post-change chlorophyll, Secchi depth, and hypolimnetic oxygen depletion rate values were estimated and compared with reported measured values for the respective characteristics after the P load reduction.

As shown in Fig. 6, the steady state load-response coupling for any given waterbody tends to move parallel to the U.S. waterbody

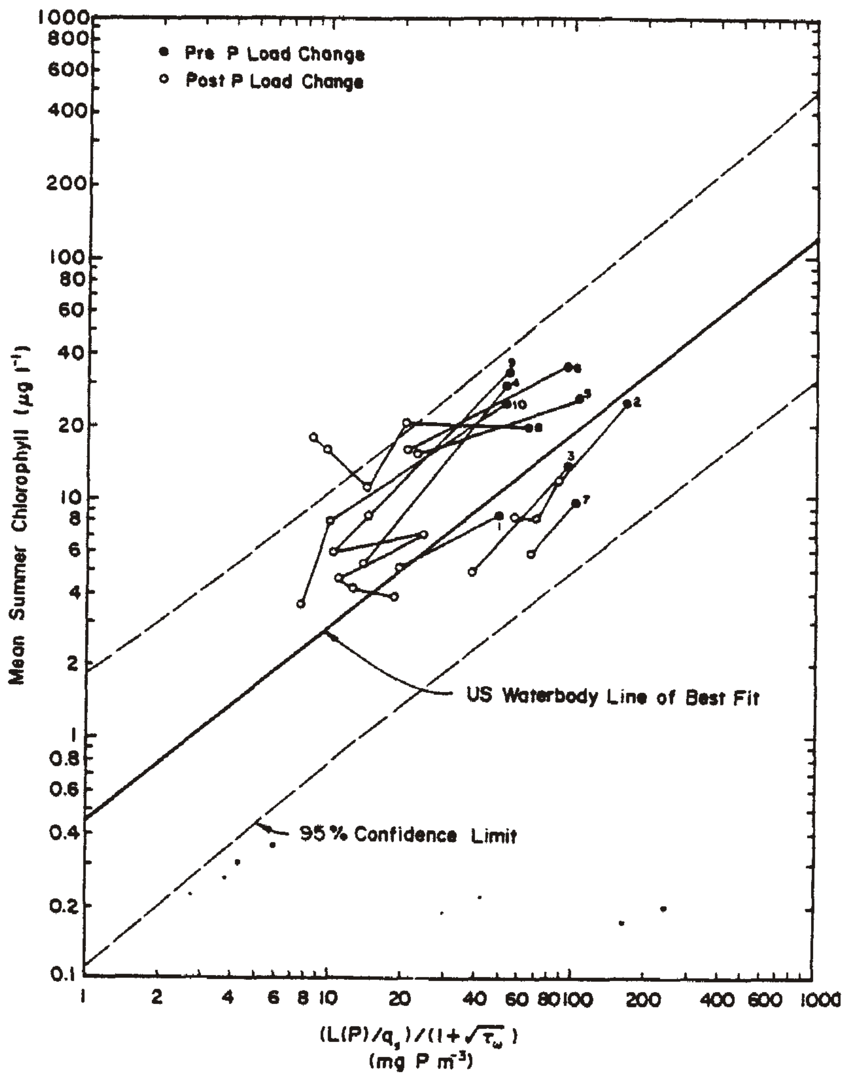



Fig. 6. Application of U.S. OECD load–response model before and after P load changes (after Rast *et al.*, 1981).

regression line. The response predicted based on a perfectly parallel response in general compared favorably with the measured values. Rest *et al.* (1981) discussed the deviations and likely reasons for them; they are related to the model constraints presented below. While the confidence intervals around the lines of best fit may appear large, this does not affect the predictive capability when one load-response coupling for a waterbody is known. It should be noted that the Vollenweider-OECD eutrophication modeling approach has an inherent verification within it since the models were developed based on the load-response characteristics of approx. 300 independent waterbodies having a variety of characteristics.

Rast *et al.* (1981) as well as Sonzogni *et al.* (1976), discussed the response time necessary to achieve the predicted steady state response to the altered P load. Provided no other substantial load changes occur, new, essentially steady state conditions will be reached after a period equal to three times the phosphorus residence time of the waterbody. They emphasized that the residence time of P, a non conservative chemical, is usually substantially shorter than that of a conservative chemical or the hydraulic residence time.

*Model constraints*

In their current forms, the OECD load-response relationships have some constraints. They are generally applicable only to waterbodies in which algal growth is limited by phosphorus. They can only be used to assess eutrophication as manifested by planktonic algal growth and have no applicability to other water quality problems such as sanitary quality, toxicants, siltation, etc. Hydraulic residence time during the growing season must be at bast 2 weeks, a load of nutrients. Further, the P load-Secchi depth relationship was developed for waterbodies having only small to moderate amounts of inorganic turbidity and color. Those waterbodies having a high inorganic particulate load or large amounts of suspended sediments or color would be expected to plot below the lines of best fit in Figs 2(B) and 4(B). The variation in the turbidity of various waterbodies plotted in Figs 2(B) and 4(B) probably accounts for much of the scatter about the lines of best fit.

There is also a number of waterbody characteristics peculiar to impoundments and some other waterbodies which must be evaluated before applying this approach, and adjusted if deemed appropriate. The depth of water in some reservoirs can be highly variable over an annual cycle. Depending on the depth-time pattern, it may be necessary, for example, to use the mean depth commonly found in early summer instead of full pool mean depth, in the models. For some reservoirs, water withdrawal is made from bottom waters. This may impact the nutrient loading since it may affect normal nutrient recycling by selectively removing hypolimnetic waters containing elevated phosphorus concentrations. Reservoirs and some lakes often have one or more arms or extensions. If these are clearly physically or chemically distinguishable from the main body of the reservoir they are likely to have discrete nutrient load-response relationships and should be considered separately. It should also be noted that these arms often trap substantial parts of the P load to the reservoir: in the case of Lake Ray Hubbard (Lee *et al.,* 1978b), one of the arms trapped about 90% of the inflowing P load. If this occurs, the P load to the main body should be adjusted accordingly. Finally, because of the nature of the systems in which reservoirs are typically constructed, many tend to contain large amounts of suspended inorganic matter. As discussed previously, such reservoirs should be evaluated to determine if the inorganic turbidity is sufficient to alter algal growth in response to nutrient load or to affect the Secchi depth measurement.

Because of the extensive number and variety of types of waterbodies to which the Vollenweider OECD modeling approach is applicable, if a waterbody does not appear to fit these relationships, it is usually an indication that either the value for one or more of the parameters has been incorrectly estimated or the users did not follow the procedures outlined by Rast & Lee (1978) of evaluating the waterbody's characteristics for any properties which would cause the nutrients to be used in a way different from the 300 waterbodies upon which the models were based (such as excessive inorganic turbidity, the presence of arms or bays which would trap nutrients, nutrient limitation, hydraulic-hydrologic characteristics, etc., as discussed previously) and making technically appropriate alterations to account for these properties. As in the case of the U.S. OECD study. for the several waterbodies which at first did not appear to have a load-response coupling similar to the other waterbodies, there were definable reasons for their behavior which could be corrected for in a technically valid manner for each waterbody based on knowledge of how the characteristic affects general nutrient utilization by planktonic algae, or by correcting errors in the data. It should be noted that such adjustments are technically appropriate to make and do not involve arbitrarily altering numbers to force the waterbody to fit. They need to be made under certain circumstances because this load-response modeling approach was developed based on theoretical nutrient utilization in completely mixed systems having a fairly simple morphology. The regressions (lines of best fit) were developed from data on waterbodies with similar "simple" characteristics or with unusual characteristics normalized. It would not be appropriate to use this approach for a waterbody having "unusual" characteristics without making such modification.

## EXAMPLE OF THE APPLICATION OF OECD EUTROPHICATION MODELING APPROACH FOR WATER QUALITY MANAGEMENT

Rast & Lee (1978) provided an extensive discussion of how this modeling approach can be used for water quality management by applying it to a "hypothetical waterbody" (Figs 7 and 8) with assigned mean depth of about 18 rn, hydraulic residence time of about 2.5 years and having about 55% of its P load from point sources (i.e. domestic wastewater inputs). These characteristics are similar to those of Lake Erie during the mid-1970s. Figure 7 shows that by reducing the point source P load to the waterbody by 90%, which can readily be done through chemical precipitation of phosphorus at domestic wastewater treatment plants, there could be some relative improvement in the recreational acceptability of this waterbody. Further improvement would be seen after reduction of the nonpoint source P load in addition to the point source control.

By using the relationships shown in Figs 2 or 4, the improvement in water quality in the hypothetical waterbody can be determined in terms of chlorophyll concentration, Secchi depth, and hypolimnetic oxygen depletion rate. Figure 8 shows, for example, that 90% point source P load reduction would cause the chlorophyll concentration to decrease from an initial 7 μg/L to about 4 μg/L; additional P load reduction from control of nonpoint sources by 40% would result in a mean chlorophyll concentration of about 3 μg/L. Changes in chlorophyll concentration on this order (i.e., from 7 to 3-4 μg/L) could be perceived by the public as improvements in water quality. Figure 8 also shows that the P load reduction that could result from detergent P bans in the watershed (i.e., 20-25% reduction of the domestic wastewater effluent P load) would not result in a perceptible change in eutrophication-related water quality in the lake.

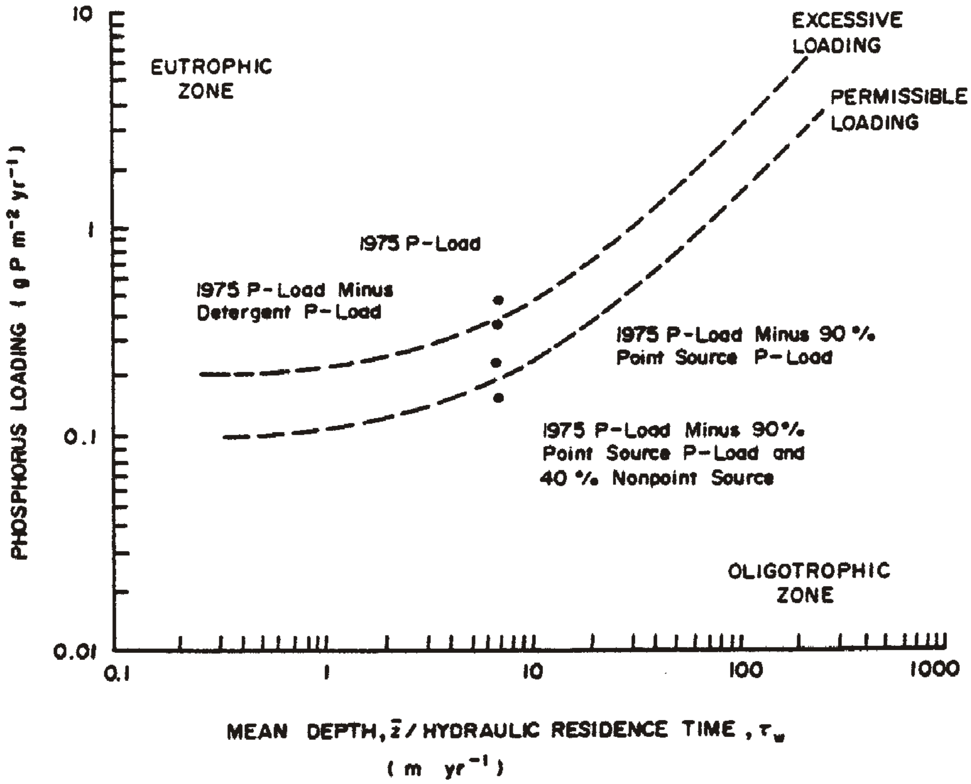



Fig. 7. "Hypothetical Waterbody" under different P load conditions applied to Vollenweider P loading–mean depth/hydraulic residence time relationship (after Rast & Lee, 1978).

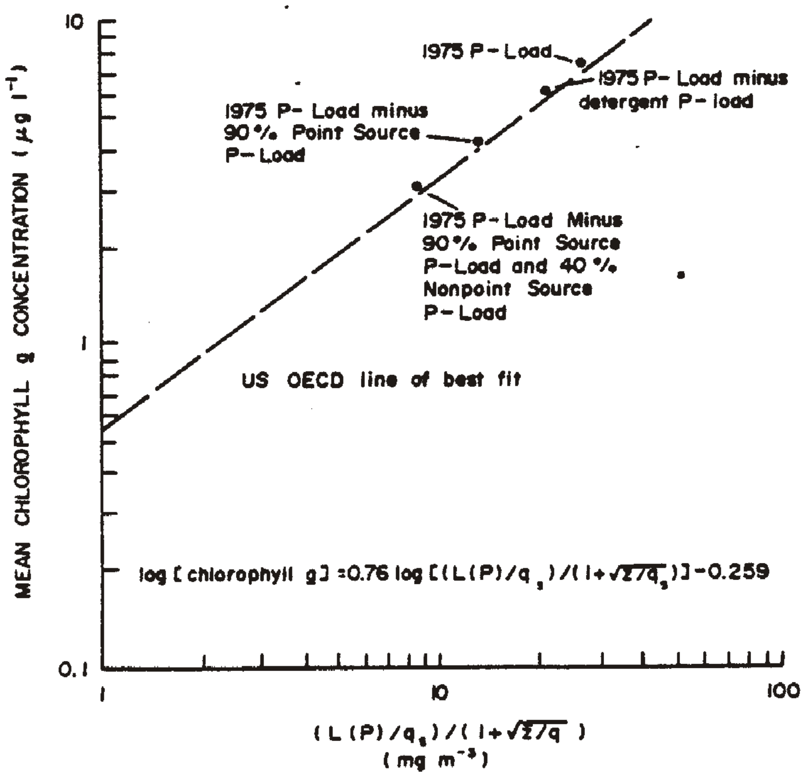



Fig. 8. U.S. OECD normalized P load–Chlorophyll concentration relationship applied to "Hypothetical Waterbody" under different P load conditions (after Rast & Lee, 1978).

The authors and their associates have made use of this Vollenweider-Rast and Lee-OECD modeling approach in eutrophication-related water quality assessment and management for a number of reservoirs in the U.S. and Europe. For example, it was used for about a dozen Tennessee Valley Authority impoundments (Newbry *et al.,* 1979, 1980); the Great Lakes (Lee *et al*., 1979); Lake Ray Hubbard, a water supply-recreation impoundment near Dallas, TX (Lee & Meckel, 1978; Lee *et al*.. 1978b); a group of waterbodies in Indiana (Lee & Archibald, 1981); Dillon Reservoir near Breckenridge, CO (Horstman *et* al., 1980); as well as the Potomac Estuary (Lee & Jones, 1981c). It is currently being used by the Spanish government under the direction of the authors, as a basis for developing a eutrophication control program for Spain's more than 700 waterbodies.

Through their work with the American Water Works Association Quality Control in Reservoirs Committee, Lee & Jones (1981a,b) and Lee *et al.* (1981a) developed a series of manuals to aid in the use of the OECD eutrophication models for water quality management. One manual describes in detail how to estimate nutrient loads to waterbodies based on land use in the watershed; another describes how to assess the limiting nutrient and factors that should be considered in making this assessment as part of a water quality management program; and the third discusses the minimum waterbody and watershed monitoring programs to collect requisite data to apply the OECD eutrophication models. Lee *et a!.* (1981b) have recently developed a step-by-step manual providing instruction on the proper application of this modeling approach for water quality management. Further, Lee *et al.* (1981c) have provided a discussion of how it can be used to comply with PL 92-500 Section 314 A lake classification-restoration requirements.

*Estimating nutrient loads*

Because of the importance of developing appropriate P load estimates in the use of the OECD (or other) eutrophication modeling approach for evaluating the efficacy of nutrient load management options as part of eutrophication management programs, a summary of key points in nutrient load estimations is presented below. Lee *et al.* (1981a) and Rast & Lee (1981) present more detailed discussions on this topic and should be consulted for further information.

In order to use the Vollenweider OECD eutrophication modeling approach to predict future water quality characteristics or to predict characteristics of impoundments which have not been constructed, it is necessary to make estimations of nutrient loadings. As discussed previously, the nutrient load to a waterbody generally consists of that derived from domestic wastewater input (point source) and that derived from nonpoint sources such as land runoff and the atmosphere. It is well established that over an annual cycle, the sediments are a sink for phosphorus (Lee *et al.,* 1977). In the U.S., the typical load of nutrient P in domestic wastewater treatment plant effluent (from plants not practicing P removal) is on the order of 1-1.5 kg P/person/year. To estimate the amounts of nutrients coming from the land and the atmosphere Rast & Lee (1978) developed, based on the literature and the U.S. OECD nutrient load and land use information, a set of nutrient export coefficients which have been found to have general applicability throughout much of the U.S. These are presented in Table 2. The export coefficient for a particular land use is multiplied by the area of land in the watershed devoted to that use, to determine the annual load from that type of land. With this information it is possible to estimate, with an adequate degree of reliability for modeling associated with management purposes, the eutrophication-related water quality impact of, for example, increased urbanization (both in terms of increased nutrient input from land runoff and increased domestic wastewater treatment plant

**Table 2. Watershed nutrient export coefficients**

| Watershed land use | Watershed export coefficient ($g\ m^{-2}\ yr^{-1}$) |
|---|---|
| | A. Total phosphorus |
| Urban | 0.1 |
| Rural/agriculture | 0.05 |
| Forest | 0.01 |
| Other: | |
| Rainfall and dry fallout | 0.02 |
| | B. Total nitrogen |
| Urban | 0.5 (0.25)* |
| Rural/agriculture | 0.5 (0.2)* |
| Forest | 0.3 (0.1)* |
| Other: | |
| Rainfall | 0.8 |
| Dry fallout | 1.6 |

* Export coefficients used in calculating nitrogen loadings for waterbodies in the western U.S.

After Rast and Lee (1978).

nutrient load) on eutrophication-related water quality; of other altered land use or agricultural practices; of phosphorus removal at domestic wastewater treatment plants (90% removal of P in the effluent can be readily accomplished by precipitation with iron or aluminum salts); of decreased P content in laundry detergents (detergent P ban could result in a 20-25% reduction in domestic wastewater treatment plant effluent P concentrations); and of the relative merits of various nutrients management strategies. It is important to note that while the Vollenweider OECD models are based on total P load, because of the model formulation and the data base used, for many waterbodies the availability of phosphorus from various sources is in large part accounted for.

## CONCLUSIONS AND RECOMMENDATIONS

The OECD Eutrophication Study nutrient load eutrophication response models provide water quality managers with a reliable, relatively easy-to-use tool for developing eutrophication-related water quality management programs. Significant advantages of the OECD modeling approach over the "dynamic" modeling approaches used or proposed for use include the facts that the OECD approach has minimal data requirements and has demonstrated predictive capability.

The Vollenweider-OECD nutrient load-eutrophication response models should be incorporated into all 314-A lake and reservoir classification and management activities. This would include establishment where necessary of a water quality monitoring program such as that outlined by Lee & Jones (1981b).

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