Alternative Approaches for Trophic State Classification for Water Quality Management

Part I: Suitability of Existing Trophic State Classification Systems
and
Part II: Application of Vollenweider-OECD Eutrophication Modeling Approach

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Part I: Suitability of Existing Trophic State Classification Systems

Abstract

Section 314-A of PL 92-500 required that each state classify the trophic status of its lakes and reservoirs and develop control programs for those judged to be excessively fertile. Part I of this two-part paper reviews approaches that have been suggested for this purpose in terms of their technical posture and the potential utility of their results. Most of the commonly-used classification schemes have several technical drawbacks. Some incorporate several parameters that address a singular water characteristic (such as total P concentration and chlorophyll both reflecting to some extent the amounts of planktonic algae present). Some include parameters that have interrelated/antagonistic implications for water quality (such as quantity of planktonic algae and aquatic macrophytes) and/or parameters that have no direct bearing on the primary production of a waterbody (such as total salts). Each of these inclusions detracts from the utility of the indexing system for water quality evaluation/management. Part II of this paper provides a technically valid and workable system that circumvents many of these problems and which the authors recommend for use in the classification of the trophic status of lakes and reservoirs.

Key Words: Eutrophication, Trophic state, Nutrients, Lakes, Reservoirs, Phosphorus

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Introduction

Section 314-A of Public Law 92-500, the 1972 amendments to the Federal Water Pollution Control Act, required that every state classify its waterbodies (lakes and impoundments) as to their degrees of eutrophication (fertilization). Further, it required that the states develop nutrient control programs for waterbodies classified as "excessively fertile." Even though this law was passed more than twenty years ago, few states have complied with both of these requirements. There are basically two reasons for this. First, as was the case with several other sections of PL 92-500, the federal Congress and the states failed to provide adequate funding for lake classification work. While the 1977 amendments to PL 92-500 (PL 95-217) made limited funds available to the states through the US EPA to begin to implement the requirements of Section 314-A, the level of funding has been inadequate to properly complete this task.

Another reason for slow compliance with PL 92-500 requirements has been an apparent lack of knowledge about how to develop an appropriate trophic state classification system and eutrophication control efficacy evaluation procedures for waterbodies. Although a number of trophic state and "restoration" evaluation schemes are available in the literature, not only has their widespread utility not been demonstrated but, as discussed below, there are significant technical deficiencies in many of the trophic state classification approaches that have been proposed for use.

While in the 1960's through the mid-1970's there was significant interest in eutrophication-related water quality evaluation and management on the part of the US EPA and its predecessor organizations, since the mid-1970's the Agency has devoted little attention to the water quality problems associated with excessive fertility of lakes, reservoirs and other waterbodies. Today, however, this situation is changing where federal and state regulatory agencies are again beginning to consider excessive fertilization of waterbodies as a major cause of water quality impairment.

The US EPA's "National Water Quality Inventory: 1992 Report to Congress" (US EPA 1994) which was released in 1994 lists aquatic plant nutrients as one of the major causes of impairment of the designated beneficial uses of the nation's waters. While there are questions about the reliability of how the US EPA and the states determined the comparative significance of aquatic plant nutrients (nitrogen and phosphorus) as a cause of impairment of the nation's waters (Lee and Jones-Lee 1994), there can be little doubt that excessive fertilization of lakes, reservoirs, estuaries and some streams and rivers is one of the dominant, if not the dominant, cause of impairment of the designated beneficial uses of waterbodies throughout the country. With federal and state regulatory agencies beginning to address point and non-point source runoff (urban and rural stormwater runoff) as a source of chemical constituents that are impairing water quality, increasing attention is being given to assessing the degree of fertility of a waterbody (trophic state) and the development of water quality standards for nitrogen and phosphorus that reflect their impact on eutrophication-related water quality.

During the 1960's for the US-Canadian Great Lakes and some Great Lakes states, considerable attention was devoted to evaluating the role of phosphorus as a cause of excessive fertilization of waterbodies. During that period and in the 1970's there were a number of research efforts devoted to evaluating the trophic state of waterbodies and the role of nitrogen and especially phosphorus in controlling this trophic state. There has been very limited research effort on this topic since the late 1970's. However, the large amount of work that was done in the 1960's and 1970's has direct applicability to developing technically valid, cost-effective approaches for evaluating the role of aquatic plant nutrients in causing excessive fertility in waterbodies and, where appropriate, in developing technically valid, cost-effective nutrient control programs. This review provides background information on the work that is pertinent to this topic area that was completed in the 1960's and 1970's that should be reviewed as part of formulating nutrient control programs.

While Section 314-A is directed toward fresh water systems, there is also growing interest in developing trophic state classification systems for estuarine and marine waters as well. McErlean and Reed (1981), for example, presented a discussion of the need for trophic classification of various parts of the Chesapeake Bay.

About 20 years ago, under the auspices of the Organization for Economic Cooperation and Development (OECD), selected investigators in the US and 17 other countries began a comprehensive study of nutrient loads and eutrophication responses in approximately 200 lakes and impoundments. The primary purpose of this investigation was to determine if it was possible to relate a waterbody's nutrient load to its eutrophication response in such a way so as to provide water quality managers with a readily usable tool for assessing the amount of nutrient control needed to achieve desired water quality - beneficial uses. Out of this study program came a series of statistical regressions between P loading and recreationally-based eutrophication-related water quality response as measured by chlorophyll concentration, Secchi depth (water clarity), and hypolimnetic oxygen depletion rate (see Rast and Lee 1978, Lee *et al.*1978a, OECD 1982). These regressions - models have demonstrated capability to predict the changes that will be brought about in these response parameters through changes in P loading (Rast *et al.*, 1983). The results of this study program provide the basis for a technically appropriate classification - evaluation system for eutrophication-related water quality management.

Presented in Part I of this paper is a general discussion of trophic status indexing schemes. It lays the groundwork for the description in Part II of this paper of an alternative approach, based on the results of the US OECD Eutrophication Study and follow-on studies, which is recommended for use in Section 314-A lake classification work in the US as well as in establishing nutrient-related water quality impairment. In addition to lacking many of the problems inherent in other indices, this approach has a demonstrated applicability to a wide variety of waterbodies.

Typical Eutrophication-Related Water Quality Concerns

The eutrophication-related characteristics of waterbodies which are usually associated with adverse impacts on beneficial uses of water include the amount and types of planktonic algae, attached algae and aquatic macrophytes; planktonic algal-related water clarity; and oxygen depletion. Planktonic algae can cause aesthetically displeasing water by decreasing water clarity, forming obnoxious growths, and causing odor problems. Algae can also interfere with water treatment for domestic use, making treatment systems less efficient and hence more costly, and/or causing taste and odor problems in the treated water. They are also implicated as a source of trihalomethane precursors (TOC and DOC) in some water supplies (Lee and Jones 1991a). With the US EPA developing regulations to limit total organic carbon (TOC) and dissolved organic carbon (DOC) in domestic water supply raw waters, increasing attention will be given to the role of algae as a source of organic carbon in surface waters. Attached algae and macrophytes can interfere with swimming, boating, waterskiing, and other recreational uses of water as well as increase the organic carbon content of water. If parts of these plants detach or break off, they can clog intake structures of water treatment facilities as well as accumulate on shorelines.

Algal and other aquatic plant growth can also result in the depletion of oxygen in the bottom waters of stratified waterbodies. Depending on the severity of the depletion, it can preclude the existence of cold water fish in waterbodies by causing a decrease in the dissolved oxygen concentrations in the cooler waters of the waterbody to levels below those tolerated by these fish (Lee & Jones 1991b). If the depletion rate is such that the hypolimnion becomes anoxic, obnoxious odors can occur in the waterbody area at overturn and during upwelling events.

Hypolimnetic oxygen depletion is also of concern to water utilities which have water supply intakes located in the hypolimnia of their supply reservoirs. As discussed by Lee and Harlin (1965), often the best raw water quality in a lake or impoundment during periods of thermal stratification is found in the hypolimnion. However, when hypolimnetic waters become anoxic they frequently contain elevated levels of sulfide, ferrous iron, and manganese II. These reduced forms of iron and manganese are oxidized during water treatment; the freshly oxidized ferric hydroxide and manganese dioxide can cause staining of clothing and facilities where the water is used, although aged $Fe(OH)_3$ and MnO_2 often do not cause such problems. At high concentrations, these chemicals can impart undesirable tastes and odors to the water.

Sulfide has an offensive odor and can, when it is oxidized, cause problems of corrosion of pipes in water treatment and distribution systems. While iron and sulfide are not released from the sediments to build up in the hypolimnia of lakes and reservoirs until the oxygen concentrations in the waters are zero, Delfino and Lee (1971) have shown that appreciable manganese release from sediments can occur while there are several mg/L dissolved oxygen (DO) present in hypolimnetic waters. Therefore, it is important for water utilities using hypolimnetic waters to watch not only for the onset of anoxic conditions in the hypolimnion, but also for low-DO conditions which could promote the release of manganese from the sediments.

Another reason for concern about hypolimnetic oxygen depletion in reservoirs is the release to downstream waters of hypolimnetic waters. Releases of anoxic waters or waters low in dissolved oxygen can adversely affect fish populations downstream. While fish may avoid the potentially hazardous areas devoid of oxygen if the discharge of anoxic waters is continuous, the periodic release of slugs of anoxic waters could readily result in the death of fish and other aquatic organisms. As discussed above, the release of waters containing sulfide and reduced forms of iron and manganese can significantly adversely affect the quality of the receiving water for use for domestic water supplies. The release of such waters can also result in a sufficiently strong sulfide odor to cause the public to avoid recreating in the area around and downstream from the dam. Krenkel *et al.* (1979) provide further information on the impact of hypolimnetic releases from reservoirs on downstream water quality.

Lee and Jones (1991b) have reviewed the impact of eutrophication on fisheries in lakes and reservoirs. They demonstrated a high correlation between phosphorus loads to waterbodies located in many parts of the world and the fish biomass present in the waterbody. While the total fish biomass increases with increasing degrees of eutrophication, the quality of the fish tends to decrease with increasing fertility. For stratified waterbodies located in temperate regions, the increased fertility leads to decreased hypolimnetic dissolved oxygen and the loss of cold-water fisheries resources. Further, as the total fish biomass increases there is a tendency for increased production of rough fish, such as carp, and stunted fish populations which are typically considered to be less desirable by the public.

In summary, eutrophication (increasing fertility) of lakes, reservoirs and many other waterbodies typically causes significant impairment of the use of waters by the public for recreational and other purposes and is significantly detrimental to domestic water supply water quality. While there may be some improvement in the desirable fish biomass in a waterbody with increased fertility, as fertility continues to increase, the increased fish biomass is often considered to be of less desirable quality.

Classical Trophic State Indexing Schemes

The classical limnological system for defining the trophic state of a waterbody has been to have an investigator rather subjectively classify the waterbody as "oligotrophic," "mesotrophic," or "eutrophic." Limnologists and water quality specialists who work with lakes tend to intuitively "know" the definition of these terms, but as discussed in Part II of this paper, until the OECD Eutrophication Study was completed there was no general consensus as to the characteristics of each of these types of waterbodies, much less a uniform or quantitative method or set of criteria for assigning waterbodies to these categories.

One of the causes for this confusion was related to differences between the purposes of limnologists and of water quality managers for characterizing or classifying waterbodies. The limnologist is largely concerned with whole-lake functioning; all major food web - energy interactions and organism functions are of interest. All major events in the waterbody such as algal blooms are of interest regardless of time of year of their occurrence and irrespective of their impact on the use of the water by people. His tendency would be to include as many ecological compartments as possible in classification indices. The interest of the water quality specialist - manager, on the other hand, is focused on the beneficial uses of the waterbody desired and designated by the public, and the timing of the potentially adverse conditions. Of paramount concern are those waterbody characteristics which describe and affect water quality - beneficial uses and their driving forces.

The significance of changes in these characteristics is judged not necessarily on the overall ecological impact, but rather on how the public responds and perceives the impact on its use of the water. For example, an early spring bloom of algae, while of interest to the limnologist, may have little water quality significance in a waterbody for which summer recreation is the designated beneficial use. In this paper, "trophic state classification" is discussed in terms of its use in water quality management; the focal point is assessing those qualities impacting beneficial uses of a particular waterbody designated by the public.

It is important to emphasize that the overall "quality" of a water is a highly subjective description. As discussed by Rast and Lee (1978), Lee *et al.* (1978a), Jones and Lee (1982a, 1986) and Lee and Jones (1988), "good" water quality does not mean the same thing to all individuals in a particular area. The successful fisherman whose primary interest is pan fish knows that those waterbodies which are green, i.e., have high planktonic algal chlorophyll levels, are generally the most productive for fish. Such water, however, may not be aesthetically pleasing to others or provide the most desirable drinking water source. Further, there are marked geographical and cultural differences among people, as well as differences in waterbody availability, which affect the public's perception of the eutrophication-related quality of water. For example, the fairly turbid waters of Lake Ray Hubbard (Lee *et al.* 1978b) (near Dallas, TX) are acceptable to many residents of that region for recreation, while they would likely be much less acceptable to people from the Upper Midwest who are accustomed to the availability of clear waterbodies.

Several reviews have been presented over the years of the various approaches that have been suggested or used for labeling - evaluating the trophic status of waterbodies. Rast and Lee (1978) presented a comprehensive review of the literature available on this topic as of the mid-1970's. They also compared the outputs of a number of the classification schemes using the US OECD Eutrophication Study data base. Wentz (1981) reviewed some of the literature on approaches that have been suggested for establishing trophic state classification systems for lakes and reservoirs. His literature review, however, is primarily a compilation of selected parts of the literature and does not include discussion of some of the papers and reports which point out the difficulties with a number of the trophic state classification systems that have been and are being used in association with eutrophication management. Lambou *et al.* (1983) compared the results of 29 classification systems in ranking the trophic statuses of 44 lakes. They reported that many of the standard approaches are not particularly effective in describing the biological manifestations of eutrophication which would affect the beneficial uses of water. A general but critical summary of trophic state indexing or classifying systems by major category is provided here.

Multiparameter Trophic Index Systems

An alternative to the oligotrophic - eutrophic classification system was originally proposed by Lueschow *et al.* (1970). They grouped a variety of nutrient load (driving force) and water quality or trophic response parameters together in a numerical ranking meant to comparatively characterize a lake's or impoundment's eutrophication-related water quality. This approach was used in the US EPA's National Eutrophication Survey (NES) (US EPA 1974) and has been used in various forms by other groups in attempting to classify the trophic status of waterbodies (Blackwell and Boland 1979, Boland 1976, Boland *et al.* 1979, Maloney 1979, Martin and Holmquist 1979, McClelland and Deininger 1981, Piwoni and Lee 1974, Taylor *et al.* 1979, Taylor *et al.* 1980, Uttormark and Wall 1975, Welby and Holman 1977, Welby *et al.* 1980). Porcella *et al.* (1980) proposed a lake evaluation index (LEI), also patterned after Lueschow *et al.*(1970) for evaluating the efficacy of lake restoration techniques, such as phosphorus load reduction, dredging, and alum treatment.

Major Problems with Multiparameter Indices

The approach developed by Lueschow *et al.* (1970) and used in various forms by others, has inherent in it a number of significant technical problems which can lead to the inappropriate classification of a waterbody's water quality-related trophic status or assessment of the degree of improvement in the beneficial uses of a waterbody that could result from a lake restoration program (Lee *et al.* 1981). One of the major problems is that both load (driving force) and response parameters are included in the index. Piwoni and Lee (1974) and Rast and Lee (1978) have pointed out that these two types of parameters should not be mixed in a water quality trophic state evaluation. A trophic state or lake evaluation index designed to assist the public and/or its representatives in assessing the cost-effectiveness and technical validity - water quality benefit of a particular activity must focus on those parameters that the public can recognize as affecting its designated beneficial uses of the water. Trophic state indices (TSI's) and LEI's should contain only those response terms relating to the beneficial use(s) of concern, if their purpose is related to water quality management.

The public is generally concerned primarily about the greenness of the water (which can be measured by planktonic algal chlorophyll), extent of macrophyte growth, water clarity - the water depth at which the bottom still can be seen (measured by Secchi depth), and the hypolimnetic oxygen depletion which affects the possibility of sustaining a cold-water fishery in a waterbody that has summer surface water temperatures greater than about 20 C, as well as the aesthetic quality (H_2S odors) and fisheries downstream of a reservoir with hypolimnetic discharges. Only those response parameters related to designated, desired beneficial uses of the waterbody should be included in a trophic state index for water quality management. For example, hypolimnetic oxygen depletion rate may not be an appropriate response parameter to consider for a waterbody used for aesthetic enjoyment or a warm-water fishery since oxygen depletion may have comparatively limited impact on these uses and thus would not be indicative of the suitability of waters for these uses. Jones and Lee (1982b) and Lee *et al.* (1995) discussed the utility of chlorophyll and Secchi depth, respectively, as water quality evaluation parameters for water supplies.

While it is typically the load and availability of phosphorus or nitrogen which control the amount of algae produced in a waterbody, inclusion in a numeric ranking of one or both of these potential driving force parameters with those describing the waterbody response to load can readily skew the index in such a way so as to cause it to have little or no relationship to the impacts of algal growth on beneficial uses of a water that the public can perceive. For example, some waterbodies have nitrate concentrations in the mg N/L range during the critical period of concern, far in excess of a growth-limiting concentration. It would therefore be technically invalid to include nitrate in the trophic state index when its concentration can change by orders of magnitude and still not affect the planktonic algal chlorophyll concentration because algal biomass is controlled by some other factor.

Including the total concentrations of the growth-limiting nutrient with response parameters, even where a relationship can be substantiated between the total concentration and algal biomass, is, at best, redundant since the former is merely a reflection of the latter. There is an inverse relationship between the concentration of available levels of the limiting nutrient and algal growth, and possibly between the non-limiting nutrients and biomass. Neither of these parameters would, therefore, be suitable for inclusion with water quality response parameters in numeric ranking systems. The Lueschow *et al.* (1970), LEI (Porcella *et al.* 1980), National Eutrophication Survey (US EPA 1974), and a variety of other trophic state indices erroneously include such parameters. Taylor *et al.* (1980) and Hern *et al.* (1981) are also critical of those trophic state classification systems which include both load and response terms and stress the importance of focusing trophic state indices associated with water quality management on response terms such as chlorophyll.

Because a regression relationship can typically be developed between the load of the limiting nutrient (driving force) and eutrophication-related water quality response, it is possible to develop trophic state indexing systems based on the driving force. However, as discussed by Piwoni and Lee (1974) and Rast and Lee (1978), if the trophic state index is to focus on loading (driving force) terms, such as nutrient loads or concentrations (which effect a eutrophication-related response), the algal growth-limiting nutrient at the time of water quality concern should be included in the index. It is generally agreed that in many parts of the US, and for that matter the world over, phosphorus is the chemical element most likely to limit maximum algal biomass production during the summer growing season in lakes and reservoirs. This does not mean, however, as is frequently assumed, that phosphorus is limiting algal growth at all times during the year.

Further, as discussed by Rast and Lee (1978), Jones and Lee (1982a, 1986) and Lee and Jones (1985), nutrient limitation during one season of the year cannot be determined by nitrogen to phosphorus ratios determined at other times of the year. It is also important to recognize that the N to P ratio, for which the value of approximately 16 to 1 on an atomic basis (7.5 to 1 on a mass basis) often used as the demarcation between N and P limitation, does not necessarily indicate what is actually limiting algal growth. It only indicates which nutrient will likely be used up first, and hence be limiting if, among other things, the supplies of the nutrients are proportionate and steady, and growth is not inhibited by some other factor such as light intensity or photoperiod. While the N to P ratio will essentially always show one or the other to be limiting (the ratio will generally be either greater than or less than 7.5 to 1), actual nutrient limitation occurs only when the concentration of the nutrient is at growth-limiting levels, which in the case of phosphorus is on the order of a few ug/L of available P (soluble orthophosphate). The growth of algae in general is not phosphorus limited if there are more than about 5 ug/L of available phosphorus in the water at the time that the sample is collected, independent of the N to P ratio. Further information on the appropriate approach for determination of nutrient limitation of a phytoplankton population is presented by Lee and Jones (1985). As noted previously, the use of a non-limiting element in a TSI or LEI can readily skew the index to the point that it has little applicability to the purpose intended.

Caution should be exercised in developing a trophic state classification which includes the use of non-available forms of nutrients, such as much of the particulate phosphorus which enters the waterbody. Lee *et al.* (1980) discussed the importance of considering available phosphorus in developing eutrophication evaluation and management programs. They and others have found that only about 20% of the particulate phosphorus derived from land runoff is available to stimulate algal growth. For a waterbody in which the dominant sources of phosphorus are particulate, such as agricultural and urban runoff, basing the trophic classification approach on total phosphorus loads will result in an erroneous classification of the trophic state of the waterbody. Such an approach could also significantly over-estimate the benefits of total phosphorus control programs on the improvement of eutrophication-related water quality that would accrue from a phosphorus control program. This is one of the major problems with the way in which the phosphorus control programs for the Chesapeake Bay system were developed.

Some trophic state indices incorporate parameters which are relatively minor, even essentially extraneous in influencing eutrophication-related water quality, such as specific conductance, as parameters upon which the trophic status is assessed. As discussed by Lee (1973), the total salt content of a waterbody (as measured by electrical conductivity) is not a reliable indicator of the trophic state of a waterbody. This parameter should not be used in a trophic state evaluation since it can change significantly without affecting planktonic algal chlorophyll levels in a waterbody to any significant extent.

In the use of most of the existing trophic state classification systems, the data for each parameter are transformed to a scale of, for example, 0 to 10 or 0 to 100, and then the scaled values are averaged in some manner. It is found, however, that the scaling offers no significant advantage in evaluating the trophic status of a waterbody or the change in beneficial uses of the water that would result from a particular lake restoration technique. In fact, this procedure can confuse the issue. Rather than incorporating a normalizing scale based on the range of values that were found, as Porcella *et al.* (1980) have done with Secchi depth, it seems more appropriate to evaluate the response parameter in terms the public can understand, such as for Secchi depth, relative to the water depth at which the bottom can be seen. Similarly, planktonic algal chlorophyll *a* is a measure of overall water "greenness," and at levels above 10 to 20 ug/L, signals the potential presence of blue-green algal scum which is usually important to the user of the lake or reservoir.

Problems with Use for In-Lake Restoration Assessment

The US EPA, through Section 314 of Public Law 92-500, was authorized to conduct demonstration projects of various lake restoration techniques including dredging and in-lake treatment with alum for co-precipitation of phosphorus. While Porcella *et al.* (1980) recommended the use of their LEI for evaluating the efficacy of such techniques, this and similar evaluation approaches have other significant technical difficulties which make their use for this purpose inadvisable. Some of these are discussed below.

Many trophic state indexing systems lack the capability to differentiate between types of beneficial use impairment because of the inclusion of often antagonistic response parameters. For example, Porcella *et al.* (1980) included both planktonic algal chlorophyll and a measure of macrophytes. The tendency in many waterbodies is for "restoration" techniques to decrease the planktonic algal populations and thus often increase the clarity of the water. Increased water clarity, however, is often followed by increased growth of attached algae and macrophytes. Canfield *et al.* (1983) discussed interrelationships between planktonic algal chlorophyll and aquatic macrophyte growth in a group of Florida lakes and provided data on the competitive nature of planktonic algae and macrophytes for available nutrients. If the beneficial use-related characteristic of concern were planktonic algae, then an index incorporating both planktonic algae and macrophytes as response parameters may indicate that no improvement resulted from the given control measure when in actuality, a noticeable improvement in desired beneficial uses may have occurred due to the increased water clarity. This discrepancy would also occur for nutrient load reduction efficacy evaluations. Similarly, dredging, when used for lake or impoundment restoration, is usually used to decrease macrophyte coverage by making the water deeper. This can also result in a trade-off of problems arising from increased phytoplankton production that may not be detected through the LEI.

Regarding the use of dredging, Lee (1973) recommended against lake-wide dredging because of the likelihood of encouraging planktonic algal-related problems. As he pointed out, if dredging is practiced, the minimum dredging necessary to make the water usable (e.g., dredging only in beach areas, areas where shoaling interferes with boat use, etc.) could have a significant impact on a waterbody's recreational acceptability to the public. Such an impact on beneficial uses, however, would not necessarily be indicated by the Porcella *et al.* (1980) LEI or other similar indices since the action would likely have limited impact on the total percentage of the waterbody having extensive macrophyte growth.

Newbry *et al.* (1981) recommended that a potentially valuable parameter for assessing the significance of macrophytes in impairing beneficial uses of a waterbody is the percentage of the waterbody having a depth of 2 m or less that is covered by macrophyte growth to a sufficient extent as to impair beneficial uses (swimming, boating, and/or fishing) in the region during the peak of macrophyte production for the summer, which is the period of high recreational use. The 2-m depth is significantly more shallow than the 10-m depth, or photic zone, recommended as the demarcation by Porcella *et al.* (1980). It is the experience of the authors that while there can be appreciable macrophyte growth at depths greater than 2 m in a lake, it is rare that this growth has a significant detrimental effect on beneficial uses of the waterbody. The profuse macrophyte growth which significantly interferes with recreational uses of a water usually occurs in areas having a water depth of 2 m or less.

Another lake restoration technique that has received considerable attention as part of the US EPA Clean Lakes Program is what is commonly called nutrient inactivation, in which alum (aluminum sulfate) is added directly to the lake. The effectiveness of this technique for controlling algae was demonstrated many years ago in Sweden; subsequently, as part of the Wisconsin Department of Natural Resources studies on lake restoration (Peterson *et al.* 1973), it was demonstrated to be successful in several lakes in Wisconsin. These studies, as well as the US EPA-sponsored studies (Cooke and Kennedy 1981), have all shown that this technique will work. However, it is a relatively expensive technique and, unlike P load control, its effect is temporary.

To properly evaluate nutrient inactivation relative to other methods of nutrient control, one has to consider not only the reduction of planktonic algal chlorophyll, improvement in Secchi depth, etc., but also the cost and the duration of the improvement in these characteristics. Nutrient inactivation may be an attractive technique if the federal government is providing a substantial part of the funding to carry out the project. However, under conditions where the local community must bear the full burden of financing restoration, it is doubtful that it will receive widespread use because of its relatively high cost and, in some instances, the relatively short persistence of effect. Any lake evaluation index which focuses only on the short-term response of a waterbody to a control measure does not properly present the total picture with respect to the overall efficacy of the control measure or its limnological impact. Therefore, if approaches such as that of Porcella *et al.* (1980) are used for evaluating the efficacy of waterbody restoration for management purposes, they should be modified to include consideration of the cost of treatment and duration of expected improved water quality - beneficial uses. Further, as noted above, they should only focus on pertinent response parameters and should not include load terms.

Other Trophic State Index Approaches

Carlson (1977) proposed a trophic state index system which is based on total phosphorus, chlorophyll, and Secchi depth. Except for the inclusion of total phosphorus as a parameter, this approach was an improvement over previously-discussed multiparameter approaches that have been used in the past. He developed a spectra of Secchi depths, and chlorophyll and P concentrations for a group of Minnesota lakes and then outlined a numerical ranking system for waterbodies based on their relative positions within these spectra. There are, however, several technical problems with his system. As discussed by Rast and Lee (1978), Carlson's index is based on a limited number of waterbodies in one geographical region of the US. It also fails to consider the beneficial uses of the waterbody being considered, how the values of the evaluation parameters affect the beneficial uses, and the public's perception of water quality. The summing of values assigned for the various response parameters has inherent in it the same problems of skewing described previously for the multiparameter indices. In addition, while Secchi depth can be a useful eutrophication response parameter, it must be used judiciously. There are situations in which inorganic turbidity - erosional material or color exert a significant control over water clarity, masking the contribution made by planktonic algae. Under these conditions it would certainly be improper to include in a trophic state indexing system, a factor for water clarity. The problems associated with using in-lake P concentrations as an indicator of water quality have been discussed previously herein and by Rast *et al.* (1983).

Uttormark and Wall (1975) developed a classification scheme for Wisconsin lakes which focused, to a greater extent than Carlson's index, on beneficial use-related responses of the waterbodies to nutrient loads. These response parameters included dissolved oxygen in the hypolimnion, Secchi depth, fish kills, and recreational use impairment. This approach has considerable merit over many of the approaches that are being used since it focuses directly on water quality problems rather than on parameters, that while possibly easy to measure, have little or no relationship to water quality impairment. However, because of the geographical limitations of their work and the importance of locale in the perception of a water's quality, application of their classification system to waterbodies outside of Wisconsin could yield misleading results.

Hakanson (1984) has attempted to develop a relationship between lake trophic level and lake sediment characteristics based on loss on ignition and nitrogen content in the sediments. The concentrations of contaminants such as nutrients in sediments are not valid indicators of the trophic conditions of the overlying water (Bortleson and Lee 1974). Further, loss on ignition is not a valid measure of organic content of lake sediments that are calcareous. Another problem with the Hakanson approach is that for many lakes, the primary source of organics in the sediments is terrestrial material and not primary production that has occurred in the waterbody.

LANDSAT Satellite Imagery

Considerable attention has been focused over the past 20 years or so on the use of remote sensing - satellite imagery to determine water quality characteristics. The LANDSAT satellite system in particular offered considerable promise for estimating eutrophication-related water quality characteristics and "trophic status" of lakes and reservoirs. Various individuals (Blackwell *et al.*1979; Boland 1976; Boland *et al.* 1979; Meinert *et al.* 1978; Welby and Holman 1977; Welby *et al.* 1980; Bagheri and Stein 1992a; Bagheri *et al.* 1993; Bagheri and Stein 1992b; Ortiz-Casas and Pena-Martinez 1989; Eckhardt and Labounty 1994; Dierberg 1991 and 1992) have investigated the use of LANDSAT imagery to estimate the trophic status of a waterbody. Several of these investigators (Blackwell *et al.* 1979; Boland 1976; Boland *et al.* 1979; Meinert *et al.*1978; Welby and Holman 1977; Welby *et al.* 1980) have reported high degrees of correlation between limnological trophic state classifications based on the multiparameter trophic classification systems of Lueschow *et al.* (1970), US EPA (1974), Porcella *et al.* (1980) and LANDSAT imagery. It appears that such correlation, however, is an artifact of the parameters used. The ability of LANDSAT to predict trophic state characteristics of lakes and impoundments for either limnological or water quality management purposes has not yet been properly evaluated. Rather than attempting to make correlations with multicomponent trophic state indices having the various inherent problems discussed in this part of the paper, LANDSAT imagery should be correlated only with individual parameters such as planktonic algal chlorophyll, extent of macrophyte coverage or Secchi depth (water clarity). These are the only three parameters that directly cause a change in response of the satellite sensors; the concentrations of chemical species such as phosphorus and nitrogen compounds which are universally used in traditional limnological trophic state indices, can change by many orders of magnitude without affecting the satellite's imagery, unless they result in changes in aquatic plant growth that is sensed by the satellite.

To properly evaluate how well satellite imagery can be related to eutrophication-related water quality characteristics of a waterbody for water quality management, it must be determined how well the satellite can estimate and differentiate between planktonic algal chlorophyll concentrations and Secchi depths in a variety of waterbodies located in various climatological, geographical, and geological regimes. The work that has been done thus far in this area does not properly evaluate how well the LANDSAT sensors can detect changes in planktonic algal chlorophyll at various chlorophyll ranges such as 0 to 5, 5 to 15, 15 to 30, and 30 to 60 ug/L chlorophyll *a*. In the higher chlorophyll concentration ranges, being able to differentiate between 5 to 10 ug/L would be adequate for most water quality-oriented trophic state classification exercises. However, where chlorophyll concentrations are less than about 10 ug/L, the satellite imagery should be accurate to within 1 to 2 ug/L. It is important that the evaluation - verification work be done with adequate frequency of measurement of the waterbody's characteristics (ground truth), following at least the minimum sampling program as recommended by Lee and Jones-Lee (1995), and in coordination with satellite passes over the areas.

One of the principal problems with LANDSAT imagery at this time in determining the planktonic algal chlorophyll in a waterbody is the amount of inorganic turbidity present in the water. At this time while LANDSAT imagery can be used to reliably determine planktonic algal chlorophyll in the open ocean, its reliability for determination of chlorophyll in near-shore marine waters, lakes and estuaries is limited. Guan *et al.* (1985) have discussed the use of the Coastal Zone Scanner on the Nimbus-7 satellite to measure chlorophyll in near-shore marine waters. The wide spectral bands of LANDSAT imagery do not allow for reliable separation of image responses from chlorophyll from those associated with inorganic turbidity (Eckhardt 1994 and Bagheri 1994). At this time considerable ground truth verification must be achieved if satellite imagery is to be used for reliable assessment of planktonic algal chlorophyll. Narrow band satellite imagery that is being developed today holds the promise of being able to distinguish between planktonic algal chlorophyll and inorganic turbidity light scattering. If this promise is fulfilled then it may be possible to establish the trophic state of waterbodies in a region with minimal ground truth through satellite imagery.

Summary

The lake evaluation indices and trophic state classification systems typically being used to comply with the mandate of the Clean Lakes section of PL 92-500 have significant technical deficiencies which do not appear to be generally recognized by the users. Many of the trophic states assigned to states' waters are biased by extraneous factors included in the classification system which have little or nothing to do with the true beneficial use-related trophic state of the waterbody. The most common problem of this type occurs in those states that have used the Carlson index in which Secchi depth was included as a parameter independent of whether or not the Secchi depth was controlled by phytoplankton, inorganic turbidity, or color. Hence, the output of these schemes are subject to significant misinterpretation in terms of which waterbodies have beneficial use-related water quality problems which are in need of nutrient control or other restoration programs, and the expected improvement in eutrophication-related water quality that could accrue from "restoration" measures.

Out of the 5-year OECD Eutrophication Study (OECD 1982) and the follow-on studies conducted by the authors in the US and elsewhere (Jones and Lee 1982a, 1986), an alternative approach for accomplishing the intent of Section 314-A of Public Law 92-500 has been developed. The technical background for, and details of this approach are discussed in Part II of this paper.

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Alternative Approach to Trophic State Classification for Water Quality Management

Part II: Application of Vollenweider-OECD Eutrophication Modeling Approach

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Abstract

Section 314-A of PL 92-500 required that each state classify the trophic status of its lakes and reservoirs and develop control programs for those judged to be excessively fertile. Part I of this two-part paper reviewed approaches that have been suggested for this purpose in terms of their technical posture and the potential utility of their results. Most of the commonly-used classification schemes have several technical drawbacks. Part II of this paper provides a technically valid and workable system that circumvents many of these problems and which the authors recommend for use in the classification of the trophic status of lakes and reservoirs. It is largely based on the results of the international OECD eutrophication studies and focuses on eutrophication response parameters of planktonic algal chlorophyll, Secchi depth and hypolimnetic oxygen depletion rates. The suggested trophic state classification approach has direct applicability to establishing nutrient-based water quality criteria for nitrogen and phosphorus compounds.

Key Words: Eutrophication, Trophic state, Nutrients, Lakes, Reservoirs, Phosphorus, Water quality criteria

Introduction

Part I of this paper presented a review of various approaches that have been used to establish the trophic state classifications of waterbodies as mandated by Section 314-A of Public Law 92-500. It also pointed out the major pitfalls that are encountered in using these systems in conjunction with water quality - beneficial use-related evaluation and management of waterbodies. Part II of this paper provides an alternative approach to satisfying the intent of PL 92-500, which circumvents many of the problems with the commonly-used indexing systems. It has as its basis the results of the US and international Organization for Economic Cooperation and Development (OECD) Eutrophication Studies and follow-on work of the authors.

About 20 years ago, under the auspices of the OECD, selected investigators in the US, Canada, Japan, Australia, and 14 European countries began a 5-year study of nutrient loads and eutrophication responses in about 200 lakes and reservoirs. The investigation focused on the quantification of the relationships between a waterbody's nutrient load and its eutrophication response that could be used by water quality managers to assess the amount of nutrient control needed to achieve desired water quality - beneficial uses. Because the US portion of this study involved the synthesis of existing data rather than new field studies, it was completed earlier than the overall study; the senior author (G. F. Lee) had the responsibility for the synthesis of the data on the nearly 40 US OECD waterbodies. Based on the data collected, and the theoretical and applied work of Vollenweider (Vollenweider 1968, 1975, 1979), a series of statistical regressions between P loading to a waterbody and eutrophication-related water quality response parameters was developed for the US OECD waterbodies (Lee *et al.* 1978a, Rast and Lee 1978). The relationships identified in the US OECD study were essentially identical to the those of the overall OECD Eutrophication Study completed several years later (OECD 1982).

Since completion of the US OECD Eutrophication Study in the mid-1970's, the authors and their associates have been able to expand on this work, evaluate the applicability of the study results under a variety of conditions, and evaluate and verify the reliability of the models in predicting the responses of waterbodies to altered P loading conditions (Jones and Lee 1982a; Jones and Lee 1986, Rast *et al.* 1983). To date, more than 750 waterbodies around the world, representing large as well as small waterbodies, lakes as well as reservoirs, shallow and deep waterbodies, highly productive as well as relatively infertile waterbodies, an Antarctic lake as well as those in which summer temperatures reach 30 to 35 C, and a variety of waterbodies in-between these extremes, have been found to respond to P loading and changes in P loading in terms of the production of planktonic algae in similar manners; these waterbodies form the data foundation for the updated Vollenweider-OECD statistical eutrophication models discussed herein.

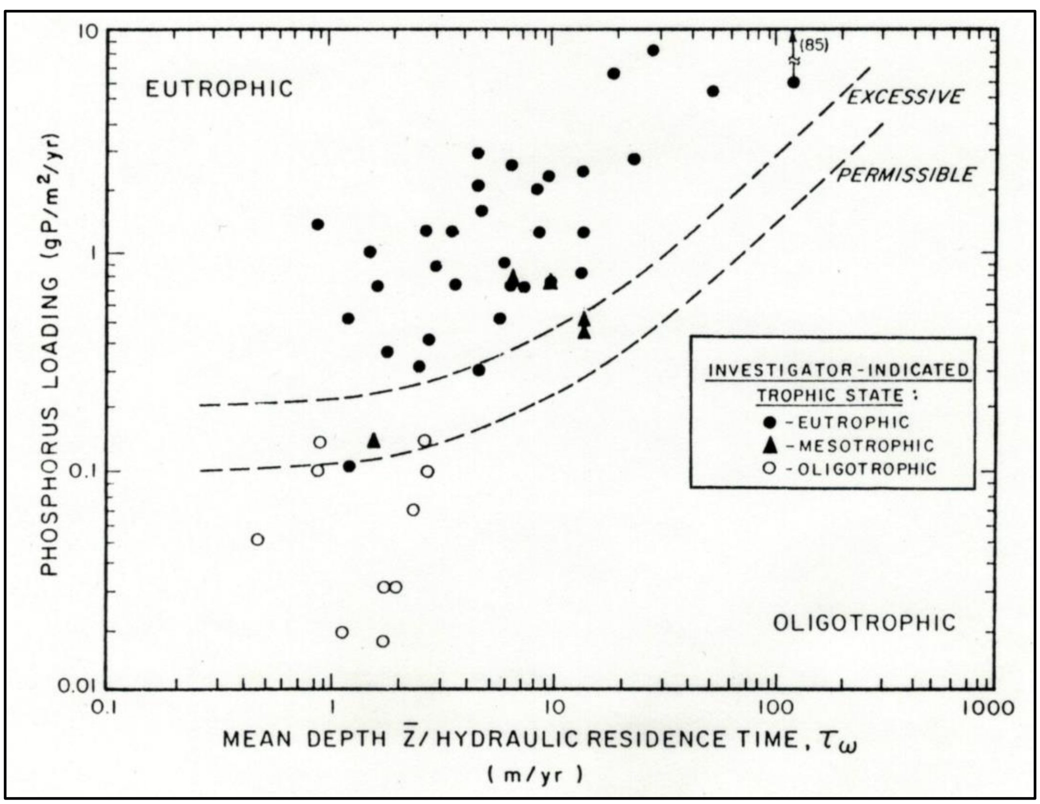
While the Vollenweider-OECD eutrophication modeling approach was not designed to serve as a trophic state indexing system per se, it can be adapted for use for identifying and evaluating imminent eutrophication-related water quality - beneficial use deterioration, and the potential improvements in the same that could be achieved by specific management options. As discussed in Part I of this paper, "trophic state" is being discussed herein as the eutrophication-related water quality characteristics as they impact the beneficial uses of a water, and not in terms of the overall ecosystem behavior and functioning. A review of the Vollenweider-OECD eutrophication modeling approach and foundation is provided below, along with a discussion of how this approach can be used in meeting the goals of Section 314-A of PL 92-500. Jones and Lee (1986) discuss in greater detail the background and development of the models.

Synopsis of Vollenweider-OECD Eutrophication Modeling Approach

Vollenweider P Loading--Mean Depth/Hydraulic Residence Time Relationship

In the late 1960's and early 1970's, Vollenweider (1975) developed a model that described a relationship between the phosphorus load to a waterbody and the relative, general acceptability of the water for recreational use. This model is shown, with the US OECD waterbodies positioned on it, in 1. Vollenweider found in applying this model (areal annual P loading plotted as a function of the ratio of mean depth to hydraulic residence time) to a group of waterbodies that waterbodies generally described as "oligotrophic" tended to cluster in one area and those which were "eutrophic" tended to plot together in another region of the . His placement of the "permissible" and "excessive" lines was based on the results of the work of Sawyer (1947) who, in the mid-1940's, studied a number of lakes in southern Wisconsin. He found that those lakes which had phosphorus concentrations at or above 10 ug P/L and nitrogen concentrations of 300 ug N/L or more at spring overturn tended to have deteriorated recreational water quality related to eutrophication during the summer months.

Figure 1: Vollenweider Normalized Phosphorus Loading -- Mean Depth/Hydraulic Residence Time Relationship for US OECD Waterbodies (after Rast and Lee, 1978)



Sawyer's assessment of "deteriorated" water quality was subjective, based on what he felt were the views of the public of southern Wisconsin in the mid-1940's on the beneficial uses of water. Vollenweider's (1975, 1976) "permissible" line corresponds to Sawyer's (1947) 10 ug P/L value, while his "excessive" line was set at twice the Sawyer value, 20 ug P/L. Vollenweider (1979), however, has emphasized that this diagram should not be used to classify waterbodies; according to him, the trophic classifications for the waterbodies he evaluated were in general given by limnologists rather than those concerned with beneficial uses of the water.

Rast and Lee (1978) found, as shown in Fig. 1, that the trophic classifications reported by the individual US OECD investigators for their respective waterbodies are in general agreement with the findings of Vollenweider; oligotrophic US OECD waterbodies generally plotted below the "permissible" line and eutrophic waterbodies tended to plot above the "excessive" phosphorus loading line. However, in the late 1970's Rast and Lee (1978) and Lee *et al.* (1978a) pointed out that some water pollution control agencies had been misusing the model shown in Fig. 1 by indicating to the public that if the P load to a waterbody that plotted just above the "excessive" line could be reduced to make it plot just below this line, then there will be a dramatic improvement in the waterbody's water quality.

These lines should not be viewed as distinct boundaries between water quality conditions. Rather, for a given abscissa position, there is a gradation of water quality with change in P load. Rast and Lee (1978) also pointed out that one should not use the lines in the Vollenweider relationship, as has been done by a number of individuals, most notably those associated with the US EPA National Eutrophication Survey (NES), to establish "critical" concentrations for excessive phosphorus loadings, without verifying for the particular waterbody that the users of the waterbody perceive the same impairment of recreational uses as the public located in southern Wisconsin in the region of the waterbodies studies by Sawyer in the early 1940's, and that recreational use is the principal beneficial use, since it was only for this use that the guidelines were developed.

The authors have encountered several situations over the past years which strongly reinforce their notion of substantial regional differences in the public's response to planktonic algal chlorophyll and Secchi depth levels. In their work with pollution control agencies and others in parts of the country outside of the Wisconsin-Michigan-Minnesota area, the authors have found almost without exception that people in these other regions do not feel that the water quality characteristics corresponding to the Sawyer-based Vollenweider "excessive" and "permissible" phosphorus loadings, are appropriate delimiters for their regions. Newbry *et al.*(1981) demonstrated this in their study of a number of Tennessee Valley Authority impoundments, as did Archibald and Lee (1981) and Lee *et al.* (1978b) for Lake Ray Hubbard, a water supply reservoir for the city of Dallas, Texas, all of which receive extensive recreational use.

In both of these areas of the country, the water can be much greener, i.e., have a much higher planktonic algal chlorophyll content, than those in southern Wisconsin and still remain acceptable for recreational use. This apparently relates to the fact that the residents near Dallas, Texas or in the Tennessee River system do not have access to as wide a variety of waterbodies with different trophic conditions to enable them to select the kind of eutrophication-related water quality that is of interest to them. Basically, the public does not seem to be as concerned about the greenness of the water if all of the accessible waterbodies in the region have the same degree of greenness. It is clear, therefore, that the Sawyer "criteria" should not be applied uniformly across the US. This finding has important implications for establishing nutrient-based water quality criteria for nitrogen and phosphorus. Basically, there should be no national single value criteria adopted as the US EPA has done for potentially toxic chemical constituents. Instead, eutrophication-related criteria for nitrogen and phosphorus should be regional, reflecting the public's experience and desire for eutrophication-related water quality.

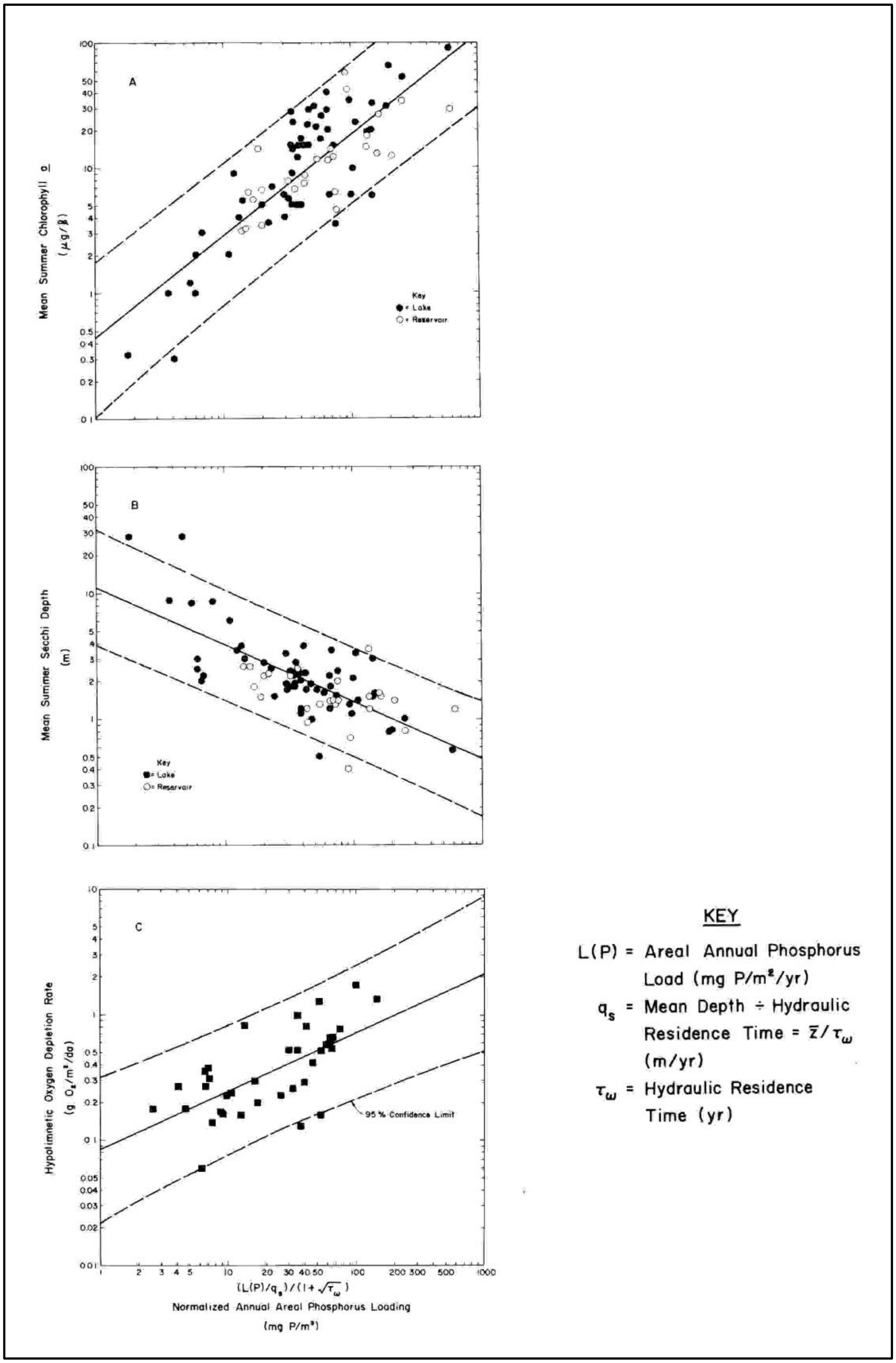
Vollenweider Normalized P Loading - Eutrophication-Related Water Quality Response Relationships

Another problem with using the model illustrated in Fig. 1 in trophic state assessment is that it is not quantitative. While as noted previously, for any mean depth/hydraulic residence time ratio (z/Tw) value water quality tends to worsen with increasing areal P load, it is difficult to determine the relationship between a waterbody's position or change in position on the graph and the degree of beneficial use impairment, or improvement in a beneficial use of the water that the public would notice. Vollenweider (1976) therefore formulated a statistical model for expressing the relationship between phosphorus load (with normalizing factors of mean depth, hydraulic residence time and waterbody area) and the planktonic algal chlorophyll concentration of waterbodies. Using the data from a set of primarily European lakes, he developed a line of best fit between the normalized P load and chlorophyll. Using the US OECD data base, Rast and Lee (1978) further demonstrated the validity of the approach, expanded the data base for and slightly revised the regression, and used the approach to develop a regression between normalized P load and Secchi depth (water clarity). Using additional data from the literature, they also developed a regression between normalized P load and hypolimnetic oxygen depletion rate. These relationships are shown in Figure. 2.

From concurrent and subsequent investigations of lakes, impoundments, estuaries, and marine waters by the authors, the load - response couplings for approximately 40 additional US waterbodies were determined (Jones and Lee 1982a) and were found to follow the US OECD regression lines. Jones and Lee (1982a) recomputed the lines of best fit for the US OECD models based on the combination of the two sets of US waterbody data and found that the updated regressions were not substantially different from the US OECD lines of best fit. Furthermore, the overall OECD data base was also found to exhibit essentially the same P load - response relationships as those found by the authors (OECD 1982).

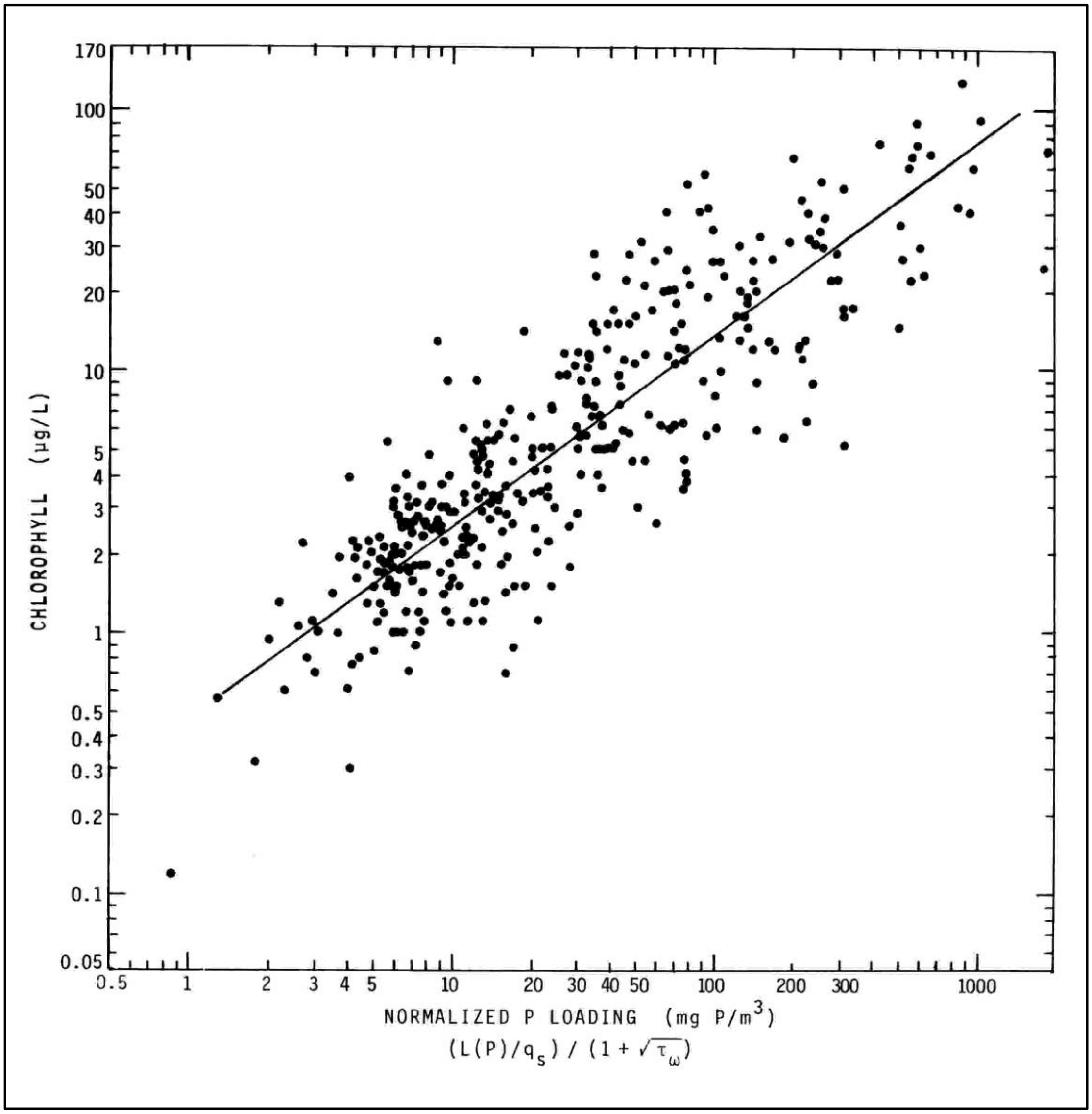
In the mid-1980's the authors integrated the results of additional normalized P load - chlorophyll response investigations on about 50 non-US waterbodies, the overall OECD data, Canadian-OECD waterbody data, and the US waterbody data to update the regression (Jones and Lee 1986). They found that their addition of the nearly 200 waterbodies to their data base for the Vollenweider-OECD model did not substantially change the regression line as expressed in Fig. 2. The updated normalized P load - planktonic algal chlorophyll relationship is shown in Fig. 3.

Figure 2: Normalized P Loading – Eutrophication-Related Water Quality Response Relationships for US Waterbodies - 1982 (after Rast and Lee, 1978)



At this time this statistical model has as its data base, the load - response couplings of more than 750 waterbodies of various characteristics from around the world.

Figure 3. Updated Vollenweider-OECD response relationship (after Jones and Lee, 1986)



While the US OECD Eutrophication Study waterbodies were generally in the northeastern, northcentral and northwestern US, the additional waterbodies incorporated by Jones and Lee (1982a) were located in other parts of the US as well. The load - response couplings for a group of Tennessee Valley Authority (TVA) impoundments (Newbry *et al*. 1981), for Lake Okeechobee, Florida (Jones and Lee 1980), and Lake Ray Hubbard as well as other waterbodies near Dallas, Texas (Archibald and Lee 1981 and Lee *et al*. 1978b), are included in the data base for the regressions. The North American Great Lakes (Lee *et al*. 1979), numerous small lakes in Indiana (Lee *et al*. 1985), as well as an alpine reservoir in the Colorado Rocky Mountains

(Horstman *et al*. 1980) are included in the Vollenweider-OECD-based load - response relationships as well.

Besides being representative of most major geographical regions of the US, and many in other parts of the world, the waterbodies that make up the regression shown in Fig. 3 have a wide variety of physical, chemical, and biological characteristics. Mean depths range from one to several hundred meters; area, from a few hectares to several thousand hectares; annual hydraulic residence times, from a few days to hundreds of years; trophic conditions, from ultra-oligotrophic to hyper-eutrophic; and climatic regimes including subtropical (with summer water temperatures in excess of 35 C), temperate (with winter ice cover several meters thick), and an Antarctic lake (Lake Vanda) that has been permanently ice-covered with 3-4 meters of ice for over 2,000 years. Lee and Jones (1991a) have found that in some years the Sacramento - San Joaquin River Delta planktonic algal chlorophyll can be estimated by the phosphorus loads to the Delta from the tributary sources. Where such estimates were not reliable, the reason for the lack of relationship in some years appears to be related to a short residence time of the nutrients in the Delta before the water is removed from it. There is insufficient time in these years to enable the algae to grow to the biomass that would be predicted based on the normalized phosphorus loads to the Delta.

The waterbodies covered include the range of alkalinity and hardness levels typically found for natural waters. While Stauffer (1985) asserted that certain Wisconsin lakes did not obey certain models that he used in examining the relationship between phosphorus loads and trophic status, his conclusions are not supported by the OECD data base where the hardness - alkalinity of the water did not affect the phosphorus load eutrophication response of the waterbodies. The waterbodies included lakes, impoundments, as well as the estuarine system of the Potomac and the main part of the Chesapeake Bay (Lee and Jones 1981). Further, Lee and Jones (1987) have found that the OECD Eutrophication Study data base-developed relationships are applicable to predicting planktonic algal chlorophyll in the New York Harbor - Northern New Jersey shore area based on the phosphorus loads from the domestic wastewaters contributed to New York Harbor from Manhattan and the Northern New Jersey shore. It is therefore apparent that the Vollenweider-OECD models are general relationships which would be expected to be applicable to many waterbodies located throughout the United States and the rest of the world.

It is important to note that contrary to the notions of some, the data requirements to apply the Vollenweider-OECD models are not extensive, and are essentially the same as those required by the Carlson (1977) trophic state index and similar approaches such as those of Larsen and Mercier (1976) and Dillon and Rigler (1974). However, for water quality management, the Vollenweider-OECD models are preferable to these others, which were in fact derived from the early Vollenweider work, because the Vollenweider-OECD model data base is considerably more extensive, covering more than 750 waterbodies of various types and degrees of fertility throughout the world. Because of their more limited scope and types of waterbodies included, these other models have built-in biases which severely limit their utility as a general approach or in evaluating eutrophication-related water quality in waterbodies outside the regions for which they were derived.

As discussed by Rast and Lee (1978) and Jones and Lee (1982a and 1986), there are waterbodies which do not fit these general relationships. Thus far these waterbodies have been easily recognized based on readily identifiable characteristics. For example, a waterbody in which the maximum summer phytoplankton biomass development is controlled by the available nitrogen may not necessarily be expected to obey phosphorus load-eutrophication response models such as the OECD eutrophication models. An example of such a waterbody is Lake Tahoe in California. Waterbodies containing large amounts of inorganic turbidity from erosion in the watershed or from suspension of the waterbody sediment would not be expected to show the same phosphorus load-eutrophication response (particularly Secchi depth) couplings as the US OECD Eutrophication Study waterbodies which, in general, had various but moderate amounts of inorganic turbidity.

Jones and Lee (1982a) and Rast *et al.* (1978) discussed how to use the Vollenweider-OECD P load - response relationships as a management tool to estimate the impact of altering phosphorus loads to a waterbody on the waterbody's eutrophication-related water quality characteristics. The P load - chlorophyll relationship shown in Figures 2 and 3 is based on average summer chlorophyll concentrations. Because, from a water quality point of view, the public is often more concerned about worst-case algal blooms, it is important to be able to estimate the magnitude of the worst algal bloom that will occur during the summer. Jones *et al.*(1979) have developed a regression based on a variety of waterbodies that shows that the worst-case summer planktonic algal chlorophyll is equal to about 1.7 times the summer mean chlorophyll. This is in general agreement with the relationship found by Vollenweider (OECD 1982) for the entire set of OECD Eutrophication Study waterbodies.

Lee and Jones (1991b) have found that the normalized P load to a waterbody is also quantifiably related to the overall waterbody fish yield (Fig. 4). In general, the greater the P load, the greater the amounts of algae and the larger the fish yield. They discuss the fact, however, that care must be exercised in focusing only on fish biomass without consideration of fish quality (e.g., type), especially when sports fisheries or commercial fisheries are of concern as desired beneficial uses of the water. While the regression line for overall fish yield in Fig. 4 is shown to be straight, if the response were "yield of desirable fish," the relationship would likely be described by three contiguous lines, each with a lower slope, as the normalized P load increased. The first change in slope would correspond to a P loading high enough to cause sufficient hypolimnetic oxygen depletion to preclude the existence of cold water fish in waterbodies having summer epilimnetic temperatures greater than about 20 C. The second change would correspond to the P loading which would promote sufficient algal growth and increase in fish biomass that the fish would become stunted. The relationships between eutrophication and fisheries are discussed in detail by Lee and Jones (1991b).

Another product of the US OECD study (Rast and Lee 1978) was the quantification of the qualitative relationship between the general recreational acceptability of water and the waterbody P load, mean depth, and hydraulic residence time shown in Fig. 1. Based on the US OECD lines of best fit for the relationships shown in Fig. 2, Rast and Lee developed the family of curves presented in Fig. 5 showing the chlorophyll, Secchi depth, and areal hypolimnetic oxygen depletion rates corresponding to each line. This provides a good illustration of the fact that

Vollenweider's "excessive" and "permissible" loading lines are not strict demarcations of water quality.

Figure 4: Relationship between Normalized P Loading and Fish Yield
(after Lee and Jones, 1991b)

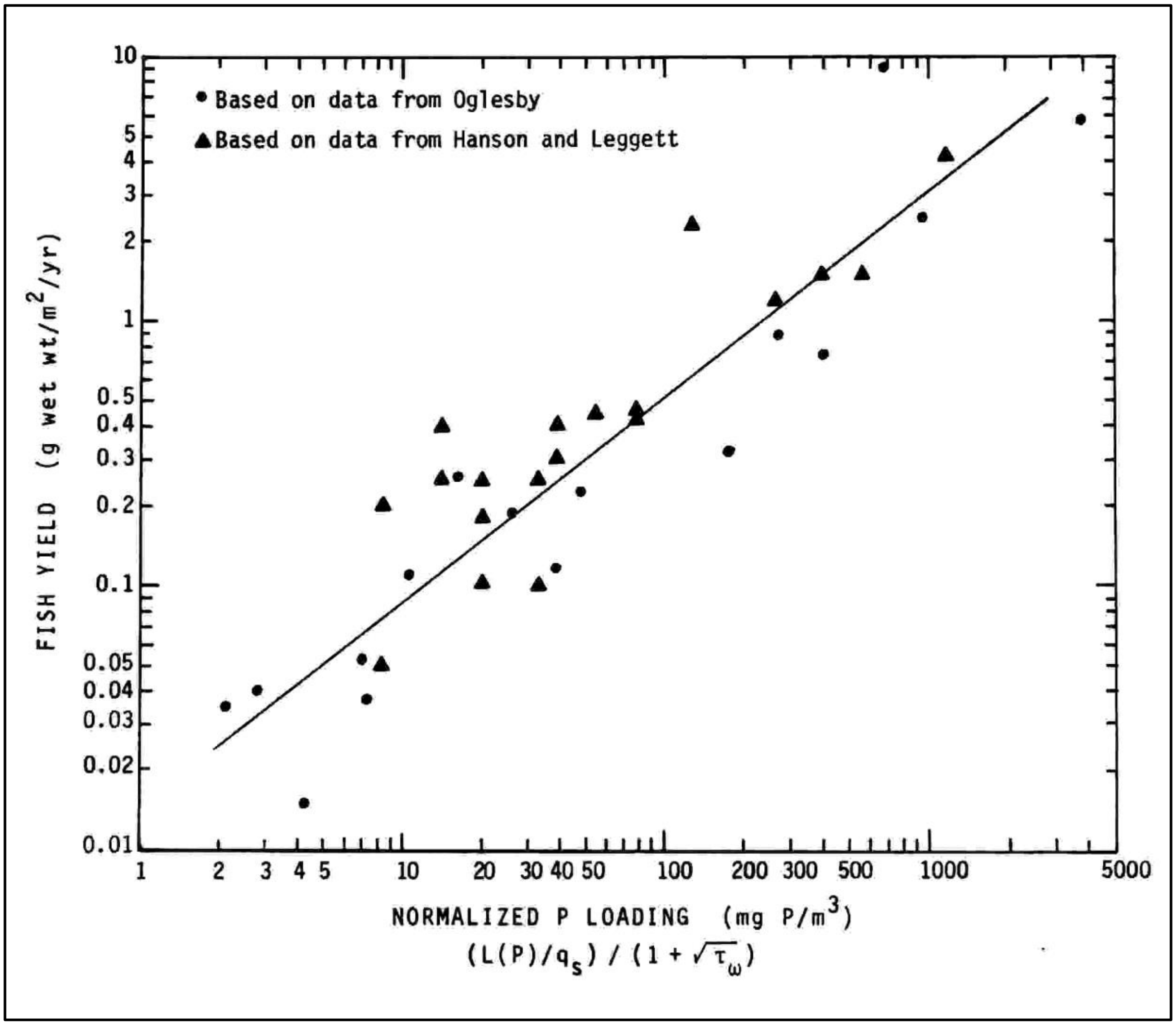
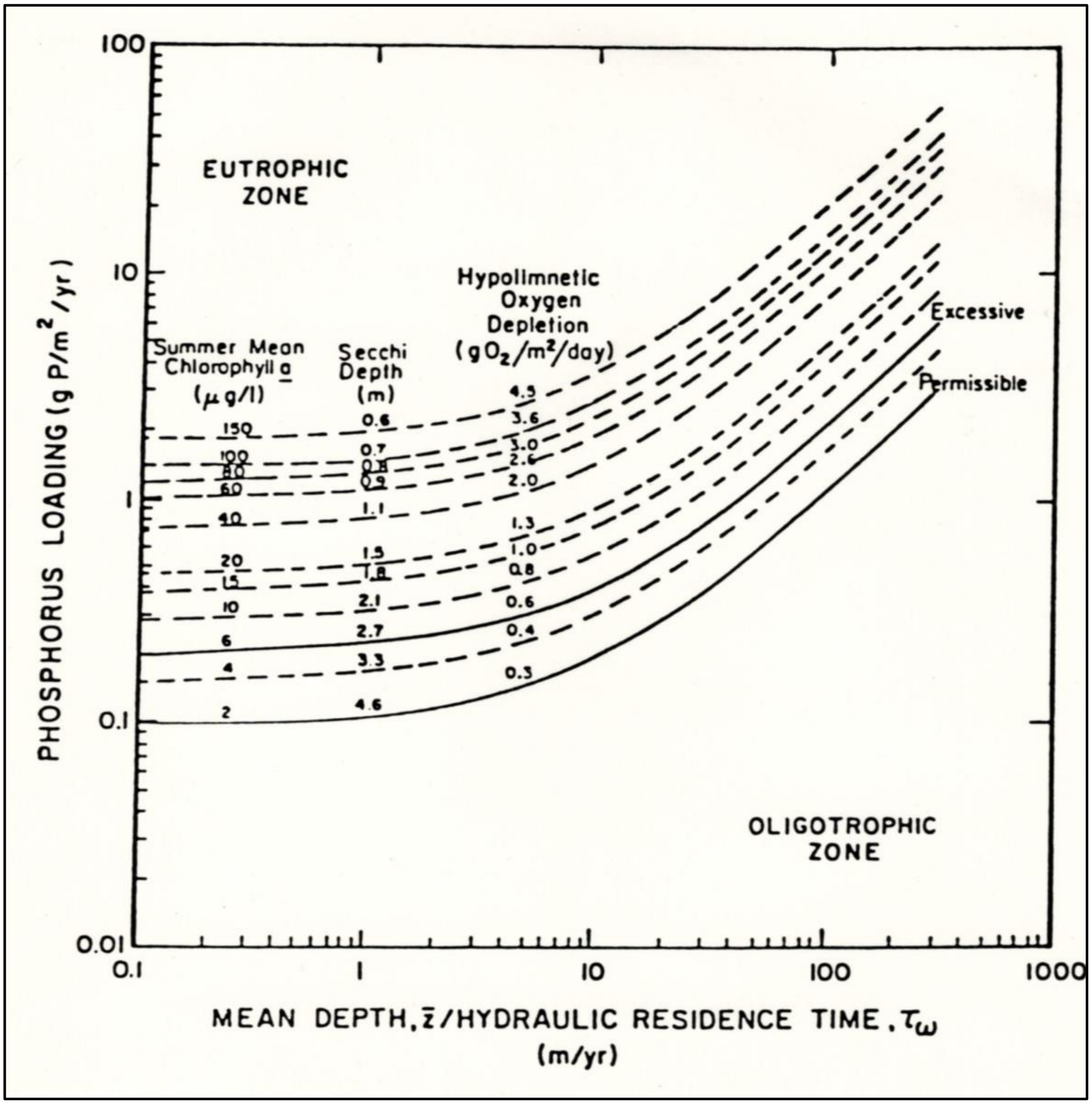


Figure 5: Quantification of Vollenweider Normalized P Loading -- Mean Depth/Hydraulic Residence Time Diagram (after Rast and Lee, 1978).

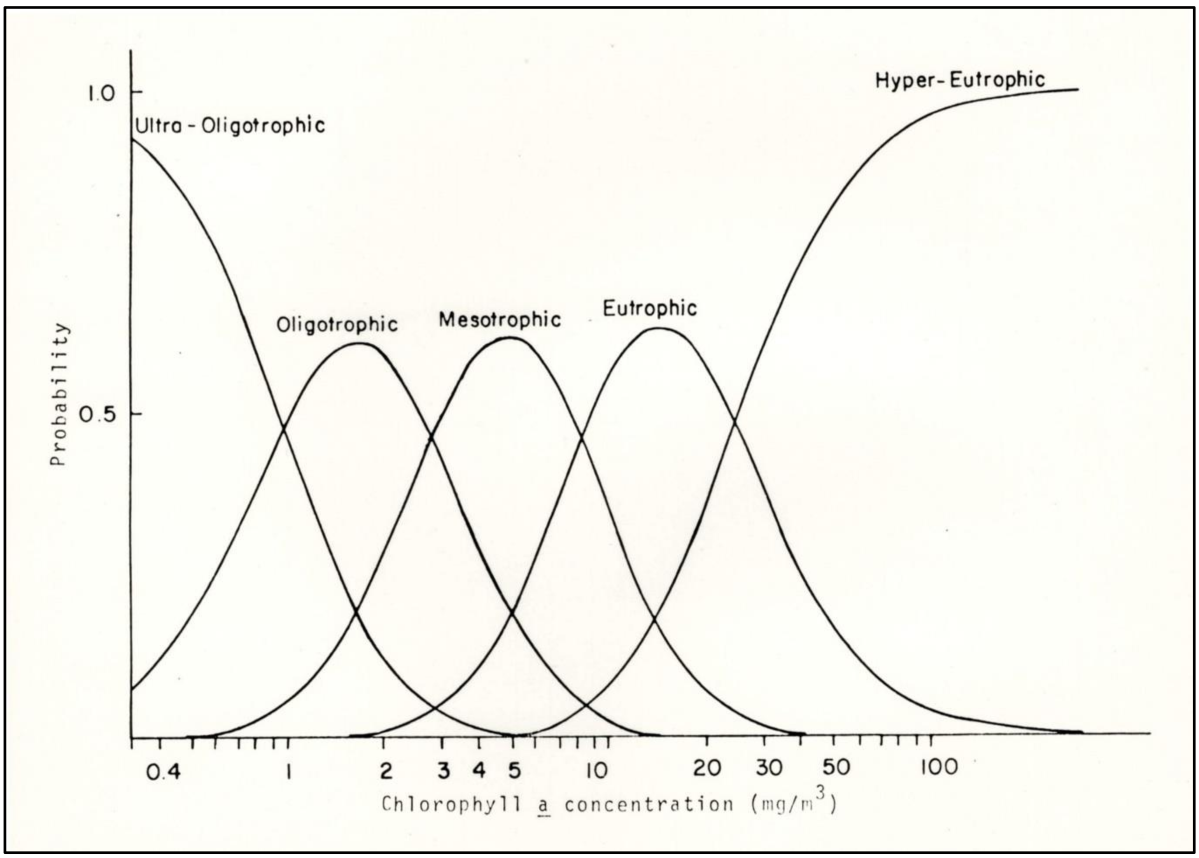


Probability Distribution for Trophic Categories

As discussed above, as part of the US OECD Eutrophication Study, Rast and Lee (1978) found reasonably good agreement between the US investigators' characterization of the "trophic status" of the waterbodies on which they had worked and the general trophic state classification suggested by the general Vollenweider load - response model (Fig. 1). Vollenweider (1976 and OECD 1982), rather than making further attempts to establish fixed numeric values to separate trophic categories, developed a probability distribution of trophic status categorization for any given chlorophyll concentration, which has as its foundation the classification of waterbodies made by the group of over 100 scientists and engineers in the fields of limnology and water quality from around the world who were involved in the international OECD Eutrophication

Study. Each investigator had complete freedom to use whatever characteristics of his/her waterbody that he/she deemed appropriate to classify it as oligotrophic, mesotrophic, or eutrophic. Vollenweider's probability distribution is shown in Fig. 6 and is read as follows: if the average chlorophyll concentration in a waterbody were, for example, 1.5 mg/m^3, there is a 60% probability that a limnologist - eutrophication water quality expert - would classify the waterbody as "oligotrophic" although there is about a 20% probability that he would classify it as "ultra-oligotrophic," or as "mesotrophic." A small but definite probability exists that a "limnologist" would classify this waterbody as "eutrophic." This, and the companion probability distributions for Secchi depth and for in-lake P concentration (OECD 1982), constitute one of the most significant contributions of the international OECD Eutrophication Study. It emphasizes the highly subjective nature of the traditional limnological trophic status classifications and illustrates the lack of discrete or absolute classes or trophic states in natural water systems even as defined by "experts" in the field.

Figure 6: Vollenweider Probability Distribution for Trophic Categories Based on Chlorophyll Concentrations (after OECD, 1982)



The probability distribution/classification scheme shown in Fig. 6 is based on the chlorophyll *a* concentration. In order to develop comparable distributions for Secchi depth specifically applicable to waterbodies in which water clarity is controlled primarily by planktonic algae, the authors established a range of chlorophyll concentrations around each peak for each major classification, and the range in-between for the intermediate (e.g., oligotrophic - mesotrophic) classifications. The normalized P loadings corresponding on the US waterbody line of best fit (Fig. 2) to the selected chlorophyll concentrations, were found. Through the normalized P loading values (Fig. 2), Secchi depths corresponding to the chlorophyll ranges were found based on the results in Fig. 2. The chlorophyll and Secchi depth values which define the limnological trophic designations based on the Vollenweider probability distribution and US waterbody lines of best fit are presented in Table 1. It is interesting to note that the 2 ug/L chlorophyll level, which corresponds to the "permissible" loading line of Figs. 1 and 5, is indicative of oligotrophic waters according to Table 1. The 6 ug/L chlorophyll level corresponding to the "excessive" loading line is indicative of mesotrophic waterbodies.

Table 1: Limnological Classification of Trophic Status of Lakes and Reservoirs

Classification	Average Planktonic Algal Chlorophyll (ug/L)	Average Secchi Depth (m)	Average In-Lake Total P (ug P/L)
Oligotrophic	< 2	> 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	>10	<1.7	>40

Also presented in Table 1 are average in-lake total phosphorus concentrations which are indicative of these categories. These values were derived from the P loading term $[(L(P)/q_s)/(1+T_w)]$. While Vollenweider (1976) had indicated that this loading term is, in theory, equivalent to the mean, in-lake, steady state P concentration [P], the results of subsequent work (OECD 1982 and Rast *et al.* 1983) on OECD and other waterbodies has shown that this identity does not precisely hold true for the data available. For the US OECD waterbodies the relationship (Rast *et al.* 1983) was: $[P] = 1.81\,[(L(P)/q_s)/(1+T_w)]^{0.81}$ which is essentially the same as the relationship found for the overall OECD Eutrophication Study waterbodies (OECD 1982). This formula was applied to the normalized P loading terms corresponding to the chlorophyll demarcations to obtain in-lake P concentrations in Table 1.

The values presented in Table 1 provide a useful guide for classification of waterbodies into traditional limnological categories. It is the experience of the authors that in such a classification, chlorophyll levels, or overall waterbody "greenness," provide the best general basis for classification for water quality management purposes. If the waterbody does not have high color or inorganic turbidity, Secchi depth (water clarity) provides the next best basis. Phosphorus concentration provides the least adequate basis for trophic state classification, especially for waterbodies in which algal growth is not P-limited or those which have large amounts of erosional inorganic material present in the watercolumn.

Total P includes soluble P and the particulate P associated with algae and inorganic particulates in the water. Where there is a substantial amount of erosional inorganic material, a significant part of the total phosphorus can be present in forms that are unavailable to support algal growth. Lee *et al.* (1980) have reported that only about 20% of particulate phosphorus present in erosional materials in the North American Great Lakes region is in a form that can be converted to algal biomass, i.e., could become available. The presence of large amounts of inorganic turbidity would impact total P concentration to a greater extent than it would Secchi depth.

Based on their review of the NES data, Hern *et al.* (1981) were also critical of the use of total phosphorus as a trophic state indicator parameter. In waterbodies that are not P-limited or that have unusually large amounts of unavailable P, there is no readily definable relationship between P concentration and algal biomass. Hence, a waterbody could be labeled as eutrophic based on the total P concentration yet have overall eutrophication response parameters more indicative of mesotrophic or oligotrophic waters. In-lake total phosphorus concentration data should only be used if other kinds of data such as planktonic algal chlorophyll or Secchi depth, are not available and then only with extreme caution and with an understanding of the limitations involved. As noted by Lee *et al.* (1995) - Part I of this paper - this reliance on P concentration as a "trophic response parameter" is one of the major drawbacks of using a number of other trophic state classification systems, such as those of Carlson (1977).

The quantitative matrix shown in Table 1 should not generally be used mechanically or exclusively for assessing eutrophication related water quality for water quality management purposes. Although the chlorophyll and Secchi depth values can generally provide insight into potential impairment of beneficial uses, phosphorus concentration alone does not; the principal aquatic plant nutrients do not impair beneficial uses of water unless their presence results in the growth of nuisance amounts of algae or macrophytes.

Not included in Table 1 is the response parameter hypolimnetic oxygen depletion rate. A number of investigators have used some measure of oxygen demand or extent of oxygen depletion as a parameter for trophic state classification for waterbodies, such as a certain degree of depletion at a particular location in the waterbody or the rate of areal depletion at a specified depth. Specifically, for example, some individuals have used the extent of oxygen depletion near the sediment/water interface as an indicator of trophic state. This approach is not valid since, as discussed by Lee and Jones-Lee (1995a), the oxygen depletion near the sediment is, for many waterbodies, controlled primarily by abiotic reactions between dissolved oxygen and reduced forms of iron and sulfur in the sediments, rather than by decomposition of algae, a reaction that would be related to trophic state. Planktonic algal-related oxygen depletion is normally manifested in a zone in the watercolumn extending from a meter or two below the thermocline to a meter or two above the sediments (Lee and Jones-Lee 1995a).

While, as shown by Rast and Lee (1978), areal hypolimnetic oxygen depletion rate in g $0_2/m^2$/day is correlated with normalized P loading, without other information about the characteristics of the waterbody, this parameter cannot be used very effectively in a limnological trophic state classification of a waterbody, much less in an assessment for water quality management. In terms of the impact of a decrease in hypolimnetic oxygen on water quality, it is usually the concentration to which it decreases, rather than the specific rate of depletion, which is of significance. For example, in some waterbodies in which surface water temperatures rise above about 20 C in the summer, the concern may be that cold water fish forced by elevated epilimnetic temperatures to live in the hypolimnion, have sufficient oxygen throughout the summer stratification period. Neither the actual amount of the change in dissolved oxygen (DO) nor the rate of change in dissolved oxygen concentration is of significance to water quality unless it results in a dissolved oxygen concentration below about 4 mg/L for sufficient periods of time to cause a significant effect on the cold water fish. A waterbody with a large hypolimnion such as Lake Ontario can maintain an oxic hypolimnion even though a waterbody with a similar algal growth pattern and oxygen depletion rate may not, because of differences in the sizes and shapes of the hypolimnia as well as climatological factors. In this regard, a waterbody can experience complete depletion of hypolimnetic oxygen during one year and not the next even though algal growth and nutrient inputs are similar, because of differences in weather patterns.

The weather will affect the data of onset of thermal stratification (which impacts the duration of stratification and the hypolimnetic temperature which affects the initial, saturated DO level), the stability and depth of the thermocline, as well as the date on which fall overturn occurs (if at all). It is therefore difficult to predict the extent to which oxygen will be depleted in a given waterbody even though the rate of depletion can be estimated. Hence it is difficult to use oxygen depletion rate alone as a parameter for describing eutrophication-related water quality or assessing the improvement in beneficial uses of a waterbody that could result from P loading changes. Because of this it is important not to use the extent of oxygen depletion in the hypolimnion, as has been done by Lueschow *et al.* (1970), Piwoni and Lee (1974), and others, as a trophic state indicator, especially for water quality management.

It is important to emphasize that the relationships and trophic status "classifications" discussed above focus on recreational uses of waters, and water characteristics which may impair these uses. If other uses are of primary importance, other characteristics or load - response couplings may be more appropriate for evaluating water quality or changes in water quality. For example, if domestic water supply is an important use for a given waterbody, then consideration must be given to response characteristics of significance to water treatment, such as length of filter runs, threshold odor number, chlorine demand, trihalomethane precursor concentration, etc. Jones and Lee (1982b) have described how water utilities can develop relationships between planktonic algal chlorophyll and raw water characteristics of importance, that are applicable to their waterbodies.

Use of Vollenweider-OECD Eutrophication Modeling Approach

As indicated previously, the development of "trophic state indices" of one type or another has, in large part, been an attempt on the part of the scientific community to provide a tool by which to implement Section 314-A of PL 92-500. It has become obvious, however, that the traditional limnological characterization of waterbodies, whether by subjective or "quantitative" means, is not appropriate for meeting the intended goal of PL 92-500, i.e., for identifying and remediation of, where necessary, impaired eutrophication-related water quality. While it is somewhat more involved and requires a degree of thought, the Vollenweider-OECD eutrophication modeling approach - specifically the updated Vollenweider-OECD load - response models - can be used to accomplish this objective. Presented below is an outline of how this can be done.

For any given waterbody, the first step is to identify the dominant beneficial uses of the waterbody. Some waterbodies will have to be segmented into areas of different primary beneficial uses; each section should be treated separately. For waterbodies having multiple beneficial uses, the first phase of evaluation should focus on the most important or most sensitive beneficial use. It would be ideal for the use-designation phase to include active public participation because the goal is to identify those uses of greatest importance to the public who ultimately pays for waterbody restoration. Beneficial uses can include water supply, sports fisheries, recreation, aesthetics, etc. There is sometimes a need to balance the quality necessary for maintaining certain multiple beneficial uses, such as a warm water fishery and aesthetically pleasing water. As discussed previously, the greater the concentration of planktonic algae, the larger the overall fish yield, but the less the aesthetic acceptability of the waterbody.

The quality or suitability of the water for the beneficial uses desired by the public can be evaluated in part by complaints received about existing water conditions or about tastes and odors in drinking water, use patterns if there are several recreational waterbodies readily accessible, etc. Park rangers and others in similar positions who have contact with the recreating public would likely be good sources of information. If there appear to be problems with the existing quality of the water, the evaluation should continue; if not, the waterbody should be classified as "currently acceptable" and handled as described subsequently.

For those waterbodies having problems of concern to the public, the most pertinent water quality response characteristics for beneficial uses should be selected. For example, for recreational use, chlorophyll and/or Secchi depth would likely be appropriate water quality indicators. At this time, the most useful response characteristic for most beneficial uses is planktonic algal chlorophyll since it, in itself, is often the problem and since it can be correlated not only with other problem characteristics but also, and most importantly, with the ultimate cause of the eutrophication-related water quality problem, namely, usually, phosphorus loading.

This discussion focuses on the most common problem of excessive fertilization of lakes and reservoirs, namely excessive growths of planktonic algae. If the phosphorus loads - eutrophication-related water quality problems - are manifested primarily as excessive macrophyte or attached algae growth, then a somewhat different approach must be used.

The next question that must be addressed for planktonic algal-related problem waterbodies is, what can be done to improve the water quality? This will require an understanding of what nutrient sources (particularly phosphorus sources) exist for waterbodies in which phosphorus either limits planktonic algal growth or is correlated with excessive planktonic algal growth, their relative importance in contributing available forms of phosphorus, which sources can be readily controlled, what it would cost to implement each control option, and most importantly, how much improvement in the beneficial uses of concern could occur as a result of adopting various possible control strategies. To make these assessments, one should begin by collecting existing data on phosphorus loading, phosphorus concentrations, chlorophyll levels, and other pertinent response characteristics. The land-use and activities of man in the watershed of the waterbody, as well as other water inputs to the system should be identified. If such information does not exist for recent years or existing information is inadequate, limited-scope water quality studies of the type described by Lee and Jones-Lee (1995b) and summarized in a subsequent section, should be conducted. The Vollenweider-OECD eutrophication modeling approach as described by Jones and Lee (1982a and 1986) should be used to integrate this information and make assessments of the potential improvements in water quality (beneficial uses) that could be attained by various management options.

The focus should be on the phosphorus loadings rather than the loadings of nitrogen, or concentrations of trace elements, etc. In the majority of lakes and impoundments, it is P which limits maximum planktonic algal biomass produced during periods of water quality concern. It is important that common pitfalls are avoided in the assessment of which nutrient is limiting planktonic algal growth in the waterbody (Lee and Jones-Lee 1995c). Further, P control options are more readily available, socially acceptable and inexpensive than other potential types of control. Even in those waterbodies or parts of waterbodies wherein P does not appear to control algal biomass, if P can be reduced to a sufficient extent, the system can be forced to P limitation and hence, responsive to P load reduction. Rast *et al.* (1983) found that waterbodies that obeyed the Vollenweider-OECD eutrophication modeling relationship for phosphorus load planktonic algal chlorophyll, where phosphorus was clearly not limiting planktonic algal growth since the concentration of available P were not decreased to growth rate limiting values, that reductions in phosphorus loads tracked in accord with those predicted based on the Vollenweider-OECD model relationships. This is an important finding in that it means that most of the waterbodies in the world will obey the phosphorus load planktonic algal relationships originally developed by Vollenweider. Further, it is discussed by Jones and Lee (1986) that it is readily possible to determine whether a waterbody would be expected to follow these relationships.

Following the procedures outlined by Jones and Lee (1982a and 1986) and briefly described above, the normalized P load - response couplings should be computed and plotted on the updated Vollenweider-OECD P load - response models (Fig. 2 or 3). Care should be taken to take into account unusual morphological/hydrologic characteristics of the waterbody (Lee and Jones 1986 and Rast *et al.* 1983) and to restrict the load - response couplings to areas of water quality concern insofar as possible. Then, based on land-use and other activities in the watershed, an evaluation should be made of the controllable phosphorus sources, the degree to which the phosphorus load from each source could be reduced, and the costs for the same. For example, a watershed may be predominantly forest with a small urban center, and have a secondary wastewater treatment plant (WWTP) that discharges to the waterbody. While typically little can be done to control the nutrients entering the waterbody from a forested area, the amounts of nutrients derived from a wastewater treatment plant can often be readily controlled. The P loading from the forest and urban areas can be estimated using phosphorus export coefficients (Rast and Lee 1983) if data are not available; the P load from the WWTP may be determined from the treatment plant records or estimated based on the size of the population served. Rast and Lee (1983) discuss in detail how these estimates can be made.

For a cost of approximately 3 to 5 cents per person per day, most secondary domestic wastewater treatment plants serving 10,000 people or more can remove 80 to 90% of the influent phosphorus, down to about 1 mg P/L in their effluents (DePinto 1980). Similarly, by altering street cleaning techniques, the total P load from urban runoff may be altered to some extent; the amount is highly dependent on localized situations and must be evaluated on a site-specific basis (Cowen and Lee 1976 and Lee 1972).

A set of estimated new phosphorus loadings for the waterbody reflecting each of the feasible phosphorus load control strategies should then be determined. These loadings can be used in the Vollenweider-OECD models (Figs. 2 and 3) to estimate the potential changes in water quality characteristics that could result from the implementation of the P control strategies once the waterbody reaches a new equilibrium with the altered P load (a period of time approximately equal to three times the P residence time); Phosphorus Residence Time = (mass of total P in lake)/(annual total P load to the lake). Estimates for the response of the waterbody (e.g., chlorophyll concentration and Secchi depth) to the altered P loading are determined to be at the intersection of a vertical line drawn through the new normalized P loading, and a line drawn parallel to the updated Vollenweider-OECD waterbody line of best fit and through the known load - response coordinate for the waterbody. As discussed by Rast *et al.* (1983), depending on the beneficial use(s) of concern, the current value of the response parameter, and the values that would result from the various P load reduction options evaluated, an assessment can be made of the improvement in beneficial uses of the water that could result from each option.

It should be noted that proper interpretation must be made of the water quality significance of given changes in a response parameter. Whether or not given change in chlorophyll concentration or Secchi depth can be perceived depends on the concentration range being considered. For example, if it is estimated that the chlorophyll concentration in a waterbody used primarily for recreational purposes will decrease from an initial concentration of 50 ug/L to 40 ug/L, many of the public would not likely see an improvement in the quality of this water. However, if the change were from an initial 12 ug/L to 5 ug/L, the improvement would likely be noticeable by essentially all of the public. Lee and Jones (1986) provide a discussion of public perception of eutrophication-related water quality. They report that at least a 20 to 25% change in the total P load to a waterbody must occur before the public would perceive a change in the planktonic algal chlorophyll for the waterbody. They found that the magnitude of this change was independent of the trophic state of the waterbody. This relationship assumed that the normal total P algal-available P relationships for most waterbodies were applicable to the waterbody of concern.

The authors and their associates have used the Vollenweider - OECD eutrophication modeling approach for evaluating the potential impact of certain P load reductions on water quality in a number of waterbodies. Horstman *et al.* (1980) determined, using this approach, that the ultraconservative, state-of-the-art P control requirements placed on domestic wastewater treatment plants in the Dillon Reservoir watershed will provide no greater improvement in the beneficial uses of this Colorado Rocky Mountain "oligotrophic" waterbody than the less conservative, more readily attainable and less expensive treatment to 1 mg P/L in the effluent. The affected communities in that region could be required to spend much more money for P removal than is necessary to maintain the desired eutrophication-related water quality. Another example is provided by Lee *et al.* (1979) who found that the reduction of phosphorus from 1 to 0.5 mg P/L in the effluent of municipal wastewater treatment plants discharging to Lakes Erie and Ontario, would have no discernible impact on the quality of the open lake water of either of these lakes. On the other hand, Lee and Jones (1980) have shown that in order to obtain desirable recreational water quality in the Platte River Power Authority's Rawhide cooling lake in Colorado, it would be necessary to reduce phosphorus in the lake's makeup waters (which are 100% secondary domestic wastewaters) to about 0.2 mg/L P.

For certain situations of excessive fertilization, consideration should be given to the use of various in-lake or localized remedial measures. Examples of situations in which such alternatives should be considered include: where there is a localized nutrient source in the vicinity of a recreational use area, where water quality concerns are in an area having limited water exchange with the main waterbody, and where the water quality problem is the presence of localized aquatic macrophyte beds. Treatment measures that should be considered, depending on the situation, include dredging, aquatic macrophyte harvesting, and in-lake alum addition. Unlike P load control, however, the impact of these measures is generally temporary, although often sufficient to significantly improve the beneficial use of the water for a period of time.

While the Vollenweider-OECD eutrophication modeling approach cannot, in general, be used to quantitatively evaluate the potential improvement in water quality that may result from these practices or to estimate how long their effect will last, Lee (1973) provides a general discussion of the applicability and effectiveness of these types of remedial measures. A literature review on lake restoration was also developed by the Wisconsin Department of Natural Resources (Dunst *et al.* 1974). The US EPA has also published several manuals (Cooke and Kennedy 1981; Pastorok *et al.* 1981; Peterson 1981 and Welch 1981) covering the experiences of investigators involved in the Clean Lakes program, with dredging, hypolimnetic aeration, in-lake phosphorus precipitation, and dilution as a method for waterbody "restoration."

Formulating Management Decisions

After all feasible management options have been explored for a waterbody which is deemed to have some eutrophication-related beneficial use impairment, and corresponding cost and impact information is available, the public should again be consulted regarding the expense it wishes to incur to achieve the projected improvements in the usability of the waterbody. The ultimate decision on lake restoration should lie with the public - the residents or users of the area and others responsible for financing such projects through park users' fees, taxes, or prices of goods.

After a decision is made regarding the eutrophication management option(s) that will be undertaken, the water quality response to that alteration should be monitored following the procedures outlined by Lee and Jones-Lee (1995b). If the management strategy involves P load alteration, the post-implementation monitoring should be carried out for a period of time equal to three times the phosphorus residence time of the waterbody (or portion thereof evaluated) plus three years (i.e., until three years beyond the new load - response steady state). If in-lake management such as aquatic macrophyte harvesting, in-lake alum addition, etc. is undertaken, the monitoring should be carried out until the duration of effectiveness of the treatment procedure is defined. At the end of the post-treatment monitoring period, the public should be consulted to determine, in the case of P load reduction, if it is satisfied with the eutrophication-related quality, or in the case of in-lake alterations, whether the improvement and duration of improvement was worth the expense and if it should be continued. As indicated previously, (Part I of this paper) it may be found that remedying one problem may either trigger or highlight another water quality problem such as reduction of planktonic algae with an increase in macrophyte production. This aspect should also be evaluated. Once a satisfactory remedy has been found for the eutrophication-related beneficial use impairment, a re-evaluation of beneficial use impairment should be conducted at about 10-year intervals.

For those waterbodies currently having no pressing eutrophication-related water quality problems based on the beneficial uses desired and designated by the public, a watershed land-use survey should be made to identify nutrient sources. Before future alterations in land use are undertaken, especially if the population in the watershed begins to increase substantially, a study program such as that outlined briefly herein and by Lee and Jones-Lee (1995b) should be undertaken to determine the potential impact that such an activity would have on the beneficial uses of the water and what preventive actions, such as P removal at domestic wastewater treatment plants, should be required.

Data Requirements for Eutrophication-Related Water Quality Assessment

One of the questions that should be addressed in connection with any "trophic state classification" or assessment as described herein, of a waterbody is, how much and what type of data should be available in order to make a classification? Some states have attempted to develop trophic state classifications based on a sampling program similar to that of the US EPA's National Eutrophication Survey (NES) (US EPA 1974). It is now clear that the NES studies, which in general involved the collection of one set of samples each spring, summer and fall, did not incorporate an adequate sampling program to properly characterize eutrophication-related water quality of the waterbodies investigated. Lee and Jones-Lee (1995b), after extensive review of the US OECD Eutrophication Study results, recommended a minimum sampling program for a waterbody and its tributaries in order to evaluate its water quality management-related "trophic state" and factors controlling its eutrophication-related water quality. As they discussed, the minimum sampling program for planktonic algal chlorophyll in a bowl-shaped waterbody should be the collection of one water sample from the middle of the waterbody at about 0.5 to 1 m depth, at preferably weekly intervals (but no less frequently than two-week intervals) during the summer growing season, which is normally the critical period for eutrophication-related water quality from the point of view of public use. These samples should be analyzed for available forms of nutrients (soluble ortho P, nitrate, ammonia), chlorophyll, etc., as outlined by Lee and Jones-Lee (1995b).

The collection of one or even three samples during the summer as was done in the NES studies, is inadequate for some waterbodies. Unfortunately, the only way to determine whether less frequent sampling can be used without losing a reliable characterization of the waterbody, is to evaluate at least one and preferably several years of data from an adequate sampling program. Long, thin lakes or reservoirs will require additional sampling locations to adequately describe the eutrophication - related characteristics of each part of the waterbody of concern; Lee and Jones-Lee (1995b) provide guidance on how such waterbodies should be sampled. To determine nutrient loading, water samples should be collected from each major tributary and each point source at least biweekly throughout the year. It is desirable to have more accurate information on major point sources if they contribute a substantial portion of the P load. It may be important to collect additional samples during heavy rains or at other times depending on the watershed characteristics, nutrient sources, and loading patterns.

The monitoring program should be carried out for at least one year, and preferably three years, to properly consider the impacts of year-to-year variation in the nutrient load and eutrophication response of waterbodies. Under no circumstances should the data from a single year's study conducted during a particularly unusual climatic year (e.g., a "wet," "dry," "hot," or "cold" year), be relied upon to characterize a waterbody. Additional details on the minimum sampling program to characterize the eutrophication-related water quality of waterbodies and to determine the primary factors responsible for determining their water quality trophic status are provided by Lee and Jones-Lee (1995b).

Suggested Approach for Trophic State Classification of Waterbodies in a Region

With the budget limitations that existed in the 1970's and, for that matter, exist today at both the federal and state levels, the 314-A coordinators in most states faced a monumental task of trying to properly classify the trophic status, i.e., eutrophication-related water quality, of all waterbodies having a surface area greater than a few tens of hectares within the state. For most states, the $100,000 provided by the US EPA was grossly inadequate to accomplish this task in a meaningful way for all waterbodies; many states have been poorly equipped from both manpower and facilities points of view to undertake this effort. Further, the US EPA provided little or no guidance on how the states should approach this mandate.

One of the most significant factors contributing to this overall problem was that during the early 1970's, the US EPA-Washington, D.C., through several administrations, essentially terminated all funding for further research and development in the field of eutrophication management. The highly effective eutrophication research group that had been developed in the 1960's at the US EPA-Corvallis, Oregon laboratory was dismantled, and inadequate funding was provided to properly complete and report on the results of the US EPA's National Eutrophication Survey and the US part of the OECD Eutrophication Study program, both of which were being conducted through this laboratory. This resulted in a significant void in the US EPA's eutrophication management capability just at the time when the greatest need was developing in the states for assistance in carrying out the provisions of Section 314-A of PL 92-500.

PL 92-500, Section 314-A required that all states develop nutrient control programs and/or adopt lake restoration techniques for all waterbodies that were found to be "excessively fertile." While the definition of "excessive fertility" was not provided in the statute, it seems reasonable to assume that excessive fertility would be defined as a degree of fertility (or amount of algae or other aquatic plants) which results in a significant impairment of a use that the public desires to make of the waterbody. Discussion was provided above on how a waterbody should be evaluated to meet the goals of PL 92-500, Section 314-A. However, in developing a program for adopting this approach, a state should give consideration to several additional points.

First, a listing of the desired beneficial uses of all waterbodies within each state or region should be compiled. Such uses as domestic water supply, recreation (boating, water skiing, fishing, swimming, etc.), agriculture, instream flow maintenance for fish, etc., should be considered. Also, those waterbodies exhibiting any impairment of existing or proposed uses should be identified as outlined previously. Of particular concern is an assessment of the public's perception of water "greenness" in impairing beneficial uses of the water, i.e., at what concentration of planktonic algal chlorophyll does the public begin to consider the water to be degraded - "polluted." This information can usually be obtained from park rangers, naturalists, and others who have frequent contact with the public who recreate in the area.

It is important if the waterbody is used for domestic water supply that the water utility be contacted concerning water treatment problems that it encounters in using the waterbody as a raw water source trying to pinpoint times and patterns of problems to the degree possible. In addition to considering use of the lake or reservoir for water supply purposes, attention should be given to downstream water uses which may be adversely affected by either surface or hypolimnetic discharges (Krenkel *et al.* 1979). As indicated previously, it is well-known that the releases of reservoirs can significantly adversely impact downstream water quality. Contaminants in a reservoir's input streams or rivers, such as iron, manganese and sulfate, which would be relatively innocuous to "downstream" users can severely impact downstream users after they pass through a reservoir system as a result of transformations of these chemicals into forms that significantly adversely impact water quality.

For those waterbodies which are determined to have impaired uses related to the degree of their fertility, the relative significance of each of the potentially significant phosphorus sources to the waterbody should be determined. Procedures such as those described by Rast and Lee (1983) for evaluating nutrient sources are usually adequate for this purpose. This should be followed by an evaluation of the controllability and impact on eutrophication-related water quality that could be attained by controlling each to a designated degree. This evaluation can readily be made using the Vollenweider-OECD eutrophication modeling approach as described previously. Once the potential improvement in water quality that would result from the various degrees of control of P from the various sources is made, cost estimates should be made for those control programs which appear to be most technically, socially, legally, and politically feasible. From this information it should be possible to formulate a management program in which the public could be informed regarding the benefits they could expect to achieve as the result of the expenditure of funds for this purpose.

It is important to note that, as discussed by Lee and Jones (1986), at least a 20 to 25% change in P load to a waterbody must occur before there will be a detectable change in eutrophication-related water quality of the waterbody. Expenditure of funds to achieve P load reductions less than this amount will not result in noticeable improvements in water quality. If the desired water quality cannot be achieved through feasible P load reductions, in-lake lake restoration techniques should be investigated.

If the above-mentioned plan is followed, it should be possible to readily ascertain whether phosphorus control and/or other lake restoration techniques would likely be effective in significantly improving eutrophication-related water quality in the waterbodies studied.

It should be emphasized that the overall "trophic state classification" approach advocated herein does not require detailed field investigations on every waterbody within a state or region of interest. Before large-scale investments are made in nutrient control, it is advisable to conduct several-year studies of the type described by Lee and Jones-Lee (1995b) to verify the nutrient loads and expected water quality response for the waterbody being considered. As indicated previously for those waterbodies with limited inorganic turbidity and color, an attempt should be made to coordinate the waterbody sampling with the passage of the LANDSAT satellite. By developing correlations between imagery and waterbody chlorophyll, Secchi depth and watershed land-use, the task of water quality evaluation in the states' other waterbodies that are judged to have eutrophication-related water quality problems will be greatly simplified. Adoption of this approach will identify those waterbodies within each state that are excessively fertile and will lead to the development of technically valid, cost-effective approaches for managing the eutrophication-related water quality.

Development of Waterbody-Specific Water Quality Criteria for Nitrogen and Phosphorus

From the relationships presented in Table 1 and Figs. 2, 4 and 5 it is possible to develop site-specific water quality criteria/standards for phosphorus in a lake or reservoir. As discussed below, the first step in the development of site-specific phosphorus criteria is the assessment of the eutrophication-related water quality of interest to those in the region of the waterbody, as expressed in planktonic algal chlorophyll, Secchi depth, hypolimnetic oxygen depletion and/or fish yield/quality. From the desired water quality it is then possible to estimate the average in-lake phosphorus concentration. From the Vollenweider-OECD relationships (OECD 1982), as well as those developed by Rast and Lee (1978) and the waterbody's mean depth and hydraulic residence time, it is possible to estimate the allowable phosphorus load to the waterbody that will achieve the desired in-lake phosphorus concentrations. This phosphorus load can be translated to concentrations of phosphorus in tributaries to the waterbody based on the discharge of each tributary. It is this approach that should be used to establish water quality criteria/standards for phosphorus. Obviously this approach should only be used for those waterbodies in which phosphorus is either limiting planktonic algal growth or where changes in phosphorus concentration can be correlated with planktonic algal chlorophyll.

While the relationships for phosphorus and eutrophication-related water quality have been reasonably well worked out for many waterbodies, similar relationships could not be developed for nitrogen. At this time there is no general relationship available that can be used to estimate the allowable nitrogen loads and in-lake nitrogen concentrations for nitrogen limited waterbodies. The OECD data base did not contain a sufficient number of nitrogen limited waterbodies to establish a reliable relationship between nitrogen loads and eutrophication response in such waterbodies. Additional work of the type that was performed during the OECD studies needs to be done for nitrogen limited waterbodies.

Lee and Jones (1988) have discussed the applicability of the Vollenweider OECD modeling results to slow-moving rivers where a plug flow model would be used to track the algal biomass development for a given phosphorus concentration in a segment of the river. As they discussed, it is not possible to establish water quality criteria for phosphorus and nitrogen as a nutrient for most stream and river systems where the nutrient impacts are manifested in attached algae and/or benthic algae. At this time there is no reliable approach for developing water quality criteria for phosphorus under these conditions.

The state of Minnesota has been in the process of developing phosphorus criteria for lakes for several years. Heiskary and Walker (1988) discuss the approach which has been adopted which involves relating the total phosphorus concentration in a lake to the public's perception of water quality problems, focusing on chlorophyll *a* and transparency as measures of problems. They utilized a user survey to assess the local public's perception of the water quality. While as discussed in Part I of this paper it is important to focus the development of phosphorus water quality criteria on response parameters, such as chlorophyll and in some waterbodies, Secchi depth where the Secchi depth is related to chlorophyll, caution should be used in correlating these parameters to in-lake total phosphorus concentrations. Since in-lake phosphorus is to some extent a driving force rather than a response parameter, erroneous assessments could develop from such a comparison.

There are waterbodies in some areas in which the total in-lake phosphorus is not related to planktonic algal chlorophyll or Secchi depth. These waterbodies would include those in which nitrogen is the controlling element governing phytoplankton production as well as those that have large amounts of inorganic turbidity. The high inorganic turbidity can result in a high total phosphorus substantial parts of which are not available to support algal growth. A more reliable approach for establishing phosphorus water quality criteria is the approach that is described herein which focuses on the development of phosphorus criteria for those waterbodies which obey the Vollenweider-OECD P load-eutrophication response relationships. The Minnesota approach does not address one of the most significant water quality problems of many small lakes and some large lakes of excessive aquatic macrophyte and attached algae growth. Neither of these would be expected to directly correlate to phosphorus loads or in-lake phosphorus concentrations. At this time there is no reliable approach to develop phosphorus criteria for these types of water quality problems.

Heiskary and Wilson (1989) and Wilson and Walker (1989) have discussed the importance of regionalizing the approach for developing phosphorus criteria. They point out that in the state of Minnesota there are markedly different ecoregions from the northern part of the state which is principally forested with large numbers of waterbodies to the southern part of the state which is agriculture with a limited number of waterbodies. These results are in accord with the authors' experience in working on lake and reservoir water quality problems throughout the US and in other countries. As discussed above, it is very important to base suitable water quality response characteristics on local conditions.

Conclusion

The international OECD and post-OECD Eutrophication Studies conducted in the 1970's coupled with the follow-on studies conducted by the authors provide an easily used, reliable technical base for developing a trophic state classification of many lakes and reservoirs. Further, these approaches enable state regulatory agencies to evaluate for those waterbodies that are classified as excessively fertile the potential benefits that can occur from controlling phosphorus loads to a waterbody to a certain degree. These same relationships can also serve as a reliable base for developing waterbody-specific water quality criteria and standards for phosphorus.

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