EVALUATION OF THE TROPHIC STATUS OF GRAND LAKE O' THE CHEROKEES WITH REFERENCE TO NUTRIENT MANAGEMENT STRATEGIES

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Bachelor of Science

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Ada, Oklahoma

1989

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
MASTER OF SCIENCE
December, 1991

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PREFACE

Global water pollution is widespread and warrants immediate attention. The causal factors of water pollution range from catastrophic events to cumulative effects of gradual perturbations. Corrective measures to these problems may require a similar range in scope of endeavors. The intent of this study is to provide new insights into the analysis of cultural eutrophication in reservoirs.

Many studies have investigated natural and cultural eutrophication of natural lakes, yet reservoir limnologists generally agree that these studies, although helpful, seldom explain the intricate mechanisms involved in eutrophication of reservoirs. Hopefully, the findings in this study combined with those of other reservoir studies will provide a better understanding of this phenomenon and stimulate research in this area. This should provide for the development of better management strategies for these valuable resources.

I gratefully acknowledge the financial support and expert advice of the personnel of the OSU-Water Quality Research Lab. I especially thank Drs. S. L. "Bud" Burks and J. L. Wilhm for providing financial assistance, professional guidance, and a comfortable environment in which to pursue my studies. Their expertise in this field has also been

very helpful.

I thank Dr. A. V. Zale for serving on my graduate committee and providing critical reviews of this manuscript. I also thank Mr. Gary Hunt of The Benham Group for providing much needed water quality data on Grand Lake. Finally, I acknowledge the graduate student body of the Department of Zoology of Oklahoma State University for their support during my professional education. Words cannot express the value of this camaraderie.

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CHAPTER I

INTRODUCTION

Many definitions and interpretations of eutrophication have been formulated. Weber (1907) originally coined the terms "eutrophic" and "oligotrophic" to describe his hypothesis of fertility of soil strata of German peat bogs. Naumann (1919) further conceptualized eutrophic and oligotrophic conditions based on characteristic phytoplanktonic associations in freshwater lakes (see Hutchinson 1957). Wetzel (1983) defines eutrophication as "nutrient enrichment" (the literal interpretation), yet continually describes eutrophication in the context of its effects; e.g., increased primary productivity and low hypolimnetic dissolved oxygen during summer (and sometimes winter) stratification. Eutrophication is hereafter referred to as nutrient enrichment.

External nutrient loads to a lake can originate from two primary sources, natural and anthropogenic activities (i.e., cultural eutrophication). Although many substances are implicated, most limnologists agree that the primary causal factors of eutrophication are increased nitrogen and phosphorus loads; albeit in many instances they do not concur on the relative importance of the individual parameters. This agreement is indicated by the widespread acceptance of Vollenweider's (1968) criteria for classifying trophic status and considerable research in the past three decades on the effects of nitrogen and phosphorus inputs on lentic ecosystems. Limnologists further agree, for reasons discussed later, that most nutrient management strategies should be directed towards reducing allochthonous phosphorus inputs.

Eutrophication is considered a natural process of lake ontogeny (Hutchinson 1957, Wetzel 1983, Henderson-Sellers and Markland 1987). The processes involved in eutrophication of freshwater lakes, especially natural lakes, have been well-researched and documented during the last three decades (Vollenweider 1968, National Academy of Sciences 1969, Likens 1972, Schindler 1974, Schindler 1977, Henderson-Sellers and Markland 1987). However, recent increasing demands on alternate freshwater resources in areas devoid of natural lakes have provided an impetus for research on artificial lakes, especially reservoirs. This research is warranted because reservoirs are more susceptible to natural ontogeny from allochthonous inputs; more afflicted by anthropogenic inputs of nitrogen and phosphorus; and serve many purposes including flood and erosion control, fisheries, recreation, and public water supplies.

Originally, the principles of previous limnological research on eutrophication of natural lakes were assumed to be comparable to those of reservoirs. However, recent studies have documented that the ontogenic processes of these two ecosystems are quite different (Thornton et al. 1990). These differences may explain many of the discrepancies resulting from the application of traditional eutrophication models/indices (e.g., Vollenweider's, Carlson's) to reservoirs.

Causal factors of eutrophication of natural lakes generally are accepted as the same in reservoirs; the differences appear to be manifested in the system's processing of these factors. One major difference in this processing is that many reservoirs exhibit "plug flow" responses to inputs instead of responding as a "completely-mixed reactor", characteristic of most natural lakes and an assumption common to most input/output models (Thornton et al. 1990). It is noted, however, that some reservoirs with long retention times, relatively small allochthonous inputs, and morphometry similar to natural lakes can be expected to exhibit more "mixed reactor" type responses. Similarly, reservoirs with extremely large or extremely small inflows, usually dictated by climatic regimes, can exhibit homogeneous properties and thus may justify use of traditional trophic state indices that operate under the "mixed-reactor" assumption. Some reservoirs, however, may fluctuate seasonally from a mixed-reactor system to a zoned plug flow system.

If reservoirs can be analyzed on the basis of their spatial and temporal heterogeneity of physical, chemical and biological activity, then a reservoir's homogeneous occurrences in time and space can be identified and subsequently evaluated with the traditional eutrophication models/indices. The purpose of this study was to elucidate an analytical technique for trophic status evaluation by using a statistical test that determines if longitudinal zonation occurs in Grand Lake O' the Cherokees, Oklahoma. Grand Lake was chosen because 1) it is highly serpentine and thus a high probability of longitudinal zonation exists, 2) water quality data from a recent (1989-1990) Clean Lakes study on Grand Lake is available, 3) the Grand River Dam Authority (GRDA) has agreed to provide additional water quality data, and 4) accelerated eutrophication in Grand Lake was recently documented (Ketelle and Uttormark 1971, EPA 1977, OSDH 1982). Additionally, when spatial and/or temporal heterogeneity prevent the use of the traditional trophic state indices, an alternative technique of trophic status evaluation was proposed and implemented.

Specifically, the objectives were to:

1. determine if longitudinal zonation in Grand Lake O' the Cherokees exists,
2. evaluate the trophic status of Grand Lake O' the Cherokees based on the findings of objectives 1 using Carlson's trophic state index before and after correction for zonation,
3. compare the congruency of these resulting trophic state indices before and after correction for zonation, and
4. estimate the restoration potential of alternative nitrogen and phosphorus reduction measures.

The statistical null hypotheses are as follows:

H_o: no longitudinal zonation exists in Grand Lake O' the Cherokees,

H_o: correction for longitudinal zonation does not improve the accuracy of trophic status evaluations using Carlson's criteria.

CHAPTER II

LITERATURE REVIEW

Eutrophication of Lakes

Definition and Implications

Eutrophication of freshwater ecosystems is a widespread problem. Lakes are thought to originate in an oligotrophic state and progress to eutrophy (Lindeman 1942). Although the process of eutrophication is a natural ontogenic process, it can be rapidly accelerated by anthropogenic input of nutrients, especially phosphorus and nitrogenous compounds. This process results in a change in the composition of the phytoplankton usually to less desirable species, (from green algae (non-diatom) and diatoms to blue-green dominance) (Schindler 1977). Blue-green algae (e.g., Aphanizomenon spp., Anabaena spp., Microcystis spp.) usually are the organisms responsible for the taste and odor problems associated with accelerated eutrophication. The filamentous forms, many of which are blue-greens, can also clog water-intake filters. The aesthetic values of a lake also decrease with accelerated eutrophication thus decreasing its public use. These problems warrant further research on the dynamics of eutrophication.

The generalized scenario in accelerated eutrophication is increased nutrients ---> increased primary productivity ---> increased organic load on sediments ---> increased oxygen demand of the sediments ---> decreased hypolimnetic oxygen ---> decreased efficiency of decomposition of organic material ---> increased sedimentation = decreased life of the lake (Wetzel 1983). This decreased lifespan of the lake decreases the effective use of the lake, which is also exacerbated by the concurrent degradation of water quality. The need to control accelerated eutrophication in reservoirs through better management strategies is intuitive. One cannot effectively manage a lake without a thorough understanding of the dynamics of eutrophication. Hence, a review of the existing knowledge and techniques for evaluation of trophic state variables is warranted.

Causal Factors

Nitrogen. All organisms require nitrogen to live. The source of nitrogen can, however, vary for different organisms. The primary source, often called the "sink", is the atmosphere. The ability to reduce atmospheric nitrogen (N_2) to form ammonia nitrogen (NH_4^+-N), nitrogen fixation, and thus make it available to biota is inherent in the blue-green algae and some heterotrophic bacteria (e.g., Azotobacter spp.) (Wetzel 1983). Nitrogen fixation by these organisms is the primary pathway by which nitrogen enters the biota in non-polluted lentic systems, albeit extensive growth of legumes in close proximity to a small waterbody can have significant effects (Wetzel 1983).

Generally, nitrogen fixation by the heterotrophic bacteria is the dominant pathway in oligotrophic waters, while nitrogen fixation in eutrophic waters is predominantly via the blue-green algae (Fig. 1). Burris et al. (1943) first associated blue-green algae with nitrogen fixation. This process is thought to occur in the heterocysts of blue-green algae and to be dependent on sunlight because vertical profiles of nitrogen fixation show similar dynamics to photosynthesis (Horne and Fogg 1970, Horne and Goldman 1972, Horne 1979).

The NH_4^+-N then can be oxidized to nitrate (NO_3^--N), a process called nitrification and predominantly restricted to the bacteria <u>Nitrosomonas</u> and <u>Nitrobacter</u> or assimilated into organic molecules (e.g., amino acids) (O'Neill and Wilkinson 1977). The organic molecules can be decomposed with the amine groups converted back to NH_4^+-N, called ammonification. The NO_3^--N can be reduced back to N_2, a process called denitrification and carried through by many bacteria, particularly <u>Escherichia</u>, <u>Pseudomonas</u>, <u>Achromobacter</u>, <u>Bacillus</u>, and <u>Micrococcus</u> (Bandurski 1965, Toetz and Cole 1980, Wetzel 1983). Alternatively, NO_3^--N can be converted to organic-N through nitrate assimilation (Keeney et al. 1971).

Although the entire nitrogen cycle is not directly implied in most eutrophication models, the important aspects and applications of the nitrogen cycle to trophic status

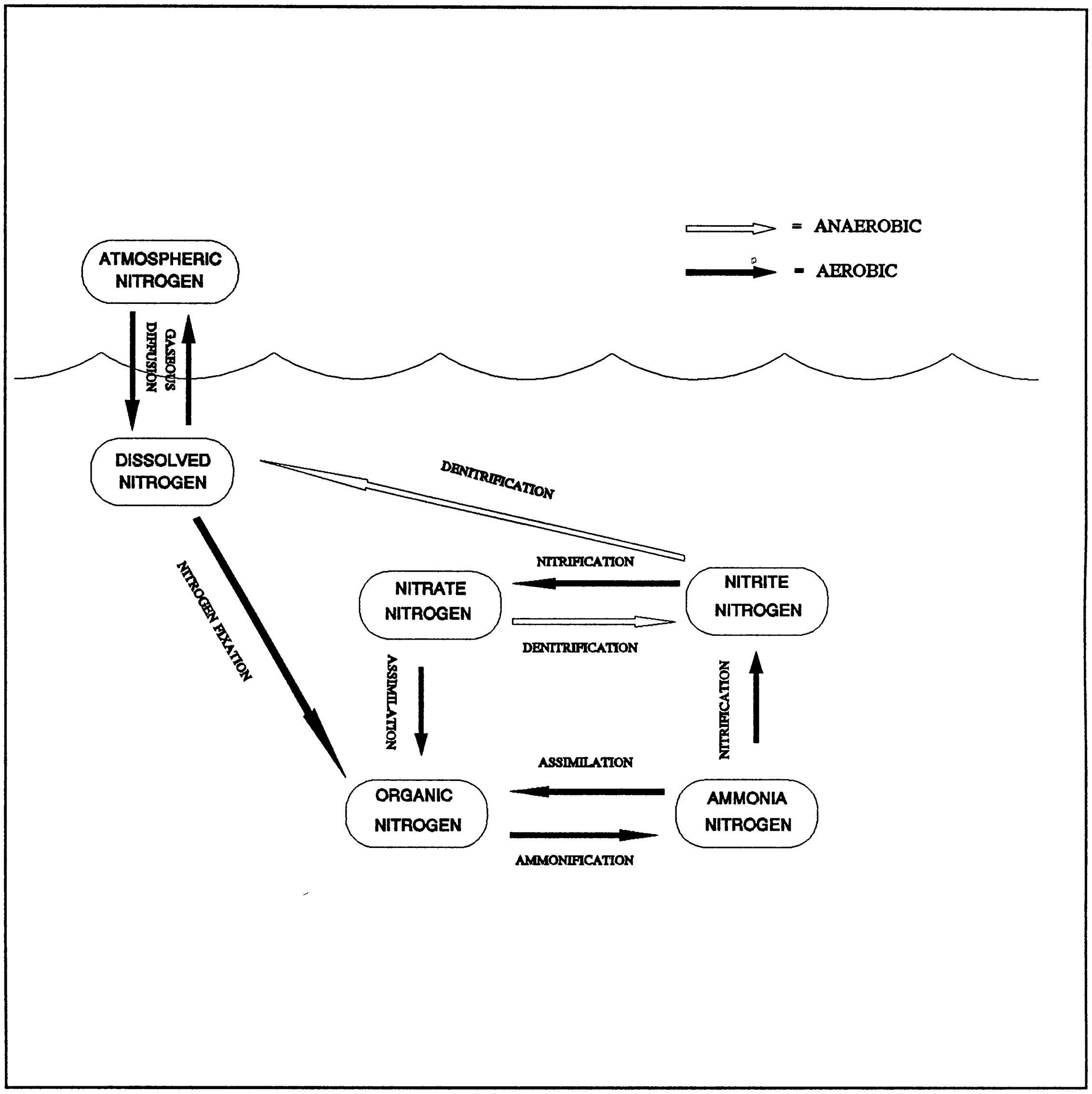

Figure 1. Nitrogen Fluxes (Taken from Reckhow and Chapra 1983).

evaluations are the following:

1. use of the ratio of NO_3^--N:NH_4^+-N as an index of sewage contamination in nitrogen enrichment eutrophication,

2. use of total nitrogen to total phosphorus ratio (TN:TP) to infer N or P limitation,

3. under nitrogen limitation (i.e., high phosphorus loads), nitrogen fixing algae have a competitive advantage and thus become dominant which can be used as a trophic state indicator (Schindler 1974).

Phosphorus. Considering all the factors implicated in accelerated eutrophication, phosphorus has been studied most extensively because it represents the greatest potential for an increase in biomass (assuming all other requirements are in excess) and has been implicated in accelerated eutrophication (Vollenweider 1968, Hutchinson 1973, Vallentyne 1974).

Forms of phosphorus in lentic systems occur in two primary fractions, inorganic and organic (Lind 1985). Lind (1985) describes several forms of each fraction based on their ease of hydrolysis. The ecological significance of these forms is still in question and hence seldom used (Wetzel and Likens 1979, Strickland and Parsons 1972). One exception is the occurrence of polyphosphates, which usually indicate anthropogenic inputs of detergents with phosphate builders (Wetzel 1983). The forms commonly used as a measure of eutrophy are total phosphorus (TP), which represents both fractions combined, and to a lesser extent orthophosphate ($\underline{o}$-PO_4^{3-}P), also called soluble reactive P (SRP) (Wetzel and Likens 1979).

The primary "natural" source of phosphorus (P) to lentic systems is the lithosphere, albeit some studies indicate significant atmospheric loads (i.e., fallout) of P in lakes whose watersheds are heavily involved in agricultural or industrial practices (Chapin and Uttormark 1973). Phosphorus occurs almost exclusively in its fully oxidized state, with the exception that some bacteria can reduce phosphates to phosphite (Hutchinson 1952, 1957).

Autochthonous inputs of phosphorus include nutrient regeneration from decomposition of planktonic organic matter and macrophytes, sediment release, and recycling by zooplankton (Stumm and Leckie 1971, Peters and Lean 1973, Lehman 1980, Wetzel 1983, Smith and Adams 1986, Graneli and Solander 1988). The dissolved phosphates from decomposition of planktonic matter and macrophytes can be transported to the trophogenic zone and rapidly assimilated by the phytoplankton. This rapid assimilation can induce significant variation in SRP and thus provides part of the rationale for using total phosphorus instead of SRP as a measure of eutrophication (Wetzel and Likens 1979).

Sediment release usually will not occur unless reducing conditions solubilize the inorganic phosphate salts (e.g., $FePO_4$) usually by reduction of its metallic counterpart (i.e., $Fe^{3+} \rightarrow Fe^{2+}$) (Wetzel 1983). In oligotrophic lakes, the oxic hypolimnion maintains a redox potential at about 200 mV or greater at the sediment/water interface which maintains maximum oxidation of the metallic ions and thus prevents formation of insoluble sulfides thereby "trapping" the phosphate salts in the sediments (Fig. 2) (Mortimer 1971). In contrast, eutrophic waters typically have anoxic hypolimnia which causes development of the reducing

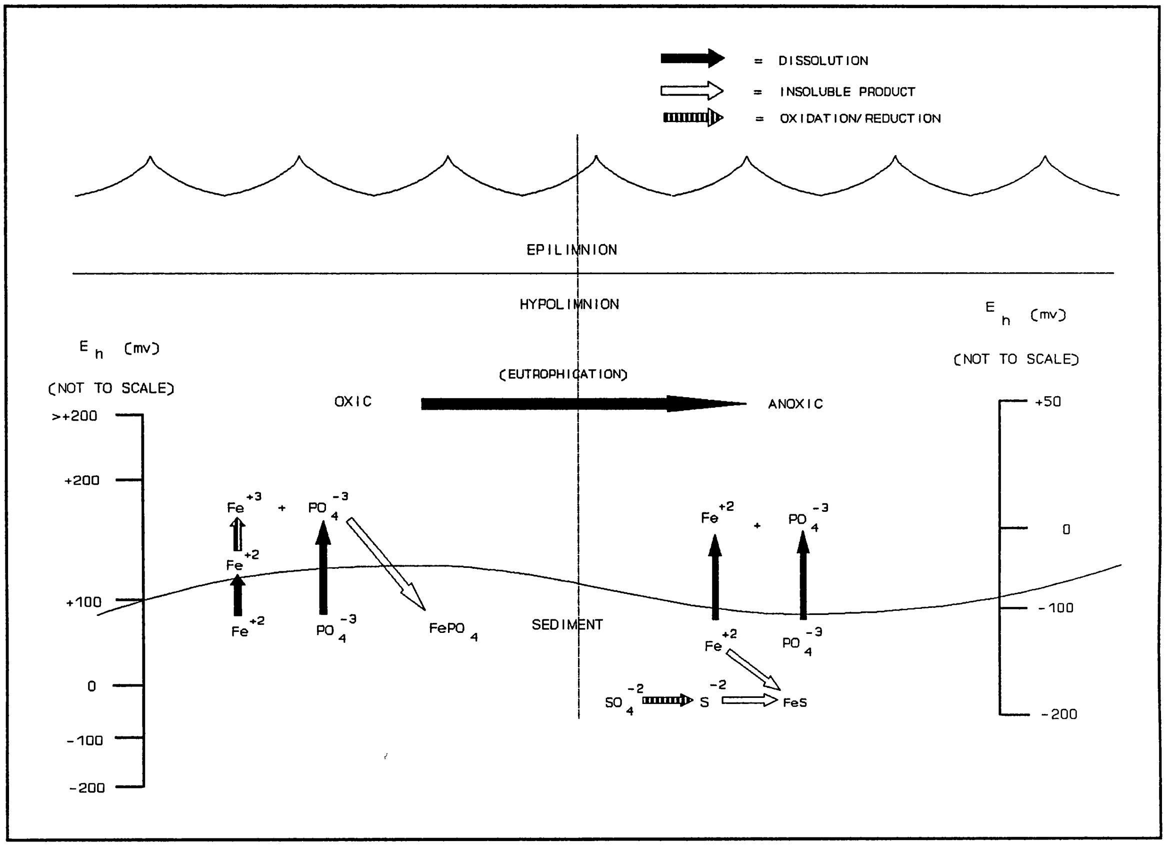

Figure 2. Mechanism of Potential Sediment Phosphorus Release.

conditions (below 100 mV) and thus promotes phosphate redissolution. This redissolution occurs by reducing the metal (e.g., Fe^{+3} --> Fe^{+2}) which in turn forms a soluble phosphate. Conversion of the insoluble ferric ion (Fe^{+3}) into a soluble ferrous ion (Fe^{+2}) shifts the equilibrium of the phosphate salt such that more phosphate is dissolved and enters the water column. The increased dissolved phosphate can accumulate and have a significant impact on epilimnetic primary productivity during autumnal mixing (Wetzel 1983).

Once the cycle has been instigated, for instance from anthropogenic inputs, accelerated eutrophication clearly can have a positive feedback effect by promoting sediment release of phosphorus which increases epilimnetic productivity which increases sediment oxygen demand upon "epilimnetic fallout", thus exacerbating hypolimnetic anoxia and perpetuating more sediment release of phosphorus.

Phosphorus released by zooplankton is rapidly assimilated by the phytoplankton (Lehman 1980). This source may also perpetuate eutrophication by increasing available P to the phytoplankton thus promoting productivity and increasing the organic load to the sediments thus increasing oxygen demand and perpetuating anoxic conditions.

Most trophic state models have been based on the dynamics of phosphorus (Vollenweider 1968, Dillon and Rigler 1974b, Larsen and Mercier 1976, Carlson 1977, Schindler 1978, Porcella et al. 1980, Canfield and Bachmann 1981, Reckhow 1988). The major reasons are because phosphorus has been found to be the limiting nutrient in most phytoplanktonic communities and is more easily contained by treatment, i.e., controlling nitrogen or carbon is virtually impossible because of their atmospheric "sinks" (Schindler 1977). By controlling phosphorus, the phytoplankton can be forced to P-limitation and thus afford the lake manager a method of controlling productivity.

Trophic State Variables

Trophic state variables include nitrogen and phosphorus loadings, productivity, organic carbon content, hypolimnetic dissolved oxygen dynamics (DO), and transparency. These factors are interrelated and, although their dynamics are similar conceptually, they differ quantitatively among lakes.

Nitrogen and Phosphorus Loadings. Nitrogen and phosphorus inputs have been shown to stimulate algal growth (Edmondson 1961, Bachmann and Jones 1974, Dillon and Rigler 1974a). These inputs, also called loads, often are standardized on an annual and areal basis, hence commonly used units are g P/m^2/yr. Phosphorus loads are dissimilar to nitrogen loads in that P-loads must be continuous to perpetuate N-limitation and thus promote long term blue-green algal dominance (Schindler 1977). The qualitative relationship between increased nutrient loads and trophic conditions has been studied in detail (Rawson 1939, 1955, Ohle 1956, Sawyer 1947, Edmondson 1961, Sakamoto 1966, Bachmann and Jones 1974, Schindler 1977). Vollenweider (1968) designated permissible N and P loadings that delineate oligotrophic, mesotrophic, and eutrophic conditions. These studies and modifications thereof have shown that increased nitrogen and phosphorus loads definitively lead to accelerated eutrophication (Vollenweider 1968, Bachmann and Jones 1974, Dillon and Rigler 1974a).

Productivity. With accelerated eutrophication, primary productivity increases (Wetzel 1983). The increase in epilimnetic standing crop causes decreased transparency, increased organic material loads on the sediment, and perpetuates the eutrophication process via the scenario previously discussed.

Organic Carbon. Subsequent to an increase in biomass, the sediment receives a larger organic carbon load which leads to a larger sediment oxygen demand. In most models, however, the organic content of the sediment as reflected in dissolved oxygen depletion is assumed to originate from epilimnetic primary productivity (Cornett and Rigler 1980, Porcella et al. 1980). In reservoirs, variable quantities ranging from little to most of the organic carbon content is derived from allochthonous inputs, especially in the upper reaches, thus total organic carbon (TOC) does not provide a reliable estimate of epilimnetic productivity in reservoirs (Thornton et al. 1990).

Hypolimnetic Dissolved Oxygen. Thienemann (1926, 1928) postulated that productivity and a lake's hypolimnetic morphometry could be predicted by the magnitude of the change in hypolimnetic dissolved oxygen. Strom (1931) and Hutchinson (1938) proposed that the morphometric parameter would interfere with oxygen consumption and thus should be eliminated by expressing the hypolimnetic oxygen depletion rate on an areal basis, called the areal hypolimnetic oxygen depletion (AHOD) rate. Hutchinson (1938) also demonstrated its validity on four lakes.

Cornett and Rigler (1980) showed that inherent assumptions in Hutchinson's model, 1) allochthonous organic matter inputs are insignificant and 2) a constant fraction of autochthonous organic matter reaches the hypolimnion, were violated in their study lakes. Cornett and Rigler (1980) further showed the AHOD rate is dependent on hypolimnetic thickness and temperature, which induced significant variation in the estimate of productivity. Although one can surmise that high AHOD rates reflect eutrophic conditions or a potential to accelerate eutrophication, its usefulness as a numerical criteria of eutrophy in reservoirs is minimal. It, however, could be used to monitor intra-lake trends (Porcella et al. 1980).

Transparency. Accelerated eutrophication decreases water transparency, commonly measured as Secchi disk depth (Wetzel 1983, Lind 1985). The consequent increase in algal standing crop can progress to the extent of decreasing the photic zone, called self-shading, and thus inhibiting further growth (Talling 1960). Carlson (1977), in developing a trophic state index, showed good correlation between a log-log transformation of chlorophyll a and Secchi disk depth (SD) ($r = 0.93$, $n = 147$) and proposed its use as a numerical criterion for trophic status.

An important assumption in this relationship is the correlation between algal standing crop and transparency. Carlson's (1977) model assumes the major contributor to transparency is algal standing crop and its respective contribution is constant, which probably is valid in most natural lakes. However, most reservoirs violate this assumption by virtue of their longitudinal heterogeneity of relative contributions of suspended sediment and algal standing crop to transparency (Thornton et al. 1990). Generally, transparency of the upper reaches of a reservoir is influenced by inflowing suspended sediment loads, while the transparency of the lower waters is related to algal standing crop (Thornton et al. 1990). Therefore, Secchi disk depths do not always correlate with the trophic status of reservoirs.

Models

The importance of phosphorus in eutrophication has provided an impetus for the development of many models that purport to predict in-lake phosphorus concentrations. When the phosphorus concentration is known or predicted, its impact on the biota is usually inferred. Phosphorus loading models have been particularly appealing to limnologists because they provide a method by which lakes and restorative measures can be assessed.

Vollenweider's Index. Vollenweider (1969) proposed the use of the mass balance equation to predict in-lake phosphorus concentration. The equation is given as:

$$V \frac{dP}{dt} = M_p - Q P - \sigma P V \tag{1}$$

where P is the in-lake phosphorus concentration (mg/l), t is time (yr), V is lake volume (m^3), M_p is the annual mass loading (g/yr), Q is the annual water loading (m^3/yr), and σ is the net sedimentation coefficient.

With the derivative of equation 1 set to zero (i.e., dP/dt = 0), steady state, and dividing through by the lake area A (m^2), solving for the in-lake phosphorus concentration gives:

$$P = \frac{L}{z_a\,(\sigma + \rho)} \tag{2}$$

where L is the annual areal phosphorus loading (g P/m^2/yr), z_a is mean depth (m), and ρ is hydraulic flushing rate (τ_w^{-1}). The four assumptions on which this model is based are:

1. σ is proportional to P,
2. the lake responds as a continuously-stirred, tank-reactor (CSTR),
3. the outflowing P concentration is equal to the in-lake concentration, and
4. the loading does not exhibit seasonality.

Dillon and Rigler's. Most of the derivations of Vollenweider's (1969) model differ in the method of estimating σ. To avoid this source of error, Dillon and Rigler (1974b) chose to work with a phosphorus retention coefficient (R) which describes the fraction of influent phosphorus that is retained by the lake. Under the above assumptions, R is given as:

$$R = 1 - \frac{\rho\ P_o}{L/z_a} \tag{3}$$

Solving for L in the Vollenweider model (2) and substituting the result into equation 3 yields:

$$R = 1 - \frac{\rho}{\rho + \sigma} \tag{4}$$

Solving for σ gives:

$$\sigma = \frac{R\ \rho}{1 - R} \tag{5}$$

Dillon and Rigler then substituted this for σ in the Vollenweider model (2) to derive:

$$P = \frac{L\ (1 - R)}{z_a\ \rho} \tag{6}$$

The P retention coefficient was experimentally calculated as the rate of mass outflow/rate of mass inflow or mathematically:

$$R_{exp} = 1 - \frac{q_o\ P_o}{\Sigma\ q_1\ P_1} \tag{7}$$

where q_o is the outflow discharge (m^3/yr), P_o is the outflow concentration (mg/l), q_i is the influent discharge (m^3/yr), and P_i is the influent concentration (mg/l). The resulting model (known as the "Dillon and Rigler" model) is:

$$P = \frac{L\,(1 - R_{exp})}{z_a\,\rho} \tag{8}$$

The study showed that this model would predict the spring P concentration in oligotrophic and mesotrophic lakes in southern Ontario with reasonable accuracy. However, Dillon and Rigler urged more studies of this nature on lakes with higher nutrient concentrations.

Kirchner and Dillon (1975) further developed the phosphorus retention model by examining the empirical relationship between R_p and other variables. They found the best multiple correlation coefficient was against areal water load (q_s), z_a, L, and the ratio of surface area to basin area (A/A_d) ($r = 0.92$). However, to reduce these data requirements for predicting R_p, Kirchner and Dillon found the best single correlation with R_p was against q_s. This relationship gave an intercept (i.e., $q_s = 0$) of $R_p = 0.75$. Since no outflow, $q_s = 0$, should reflect full retention, $R_p = 1$, a double exponential model with an intercept of 1 was implemented and gave:

$$R_p = 0.426\, e^{-0\ 271\, q_s} + 0.574\, e^{-0\ 00949\, q_s} \tag{9}$$

A comparison of the predicted R_p to the measured R_p for 15 southern Ontario lakes showed good agreement (r = 0.94). Kirchner and Dillon could not explain, however, why the best predictor variable was the areal water load instead of the volumetric load.

Chapra (1975) attempted to explain this phenomenon by using the mass balance equation as:

$$V \frac{dP}{dt} = W - Q P - K V P \qquad (10)$$

where W is the rate of mass input of phosphorus and K is the first order rate constant for the loss of phosphorus to the sediments (Vollenweider's σ) and R_p as the ratio of the rate of P lost by sedimentation to the rate of mass input or mathematically:

$$R_p = \frac{K V P}{W} \qquad (11)$$

When the steady state solution for the phosphorus concentration of the mass balance equation 10 was substituted in equation 11 the resulting R_p was given as:

$$R_p = \frac{K}{\rho + K} \qquad (12)$$

where ρ was the volumetric water load (Q/V). However, if the phosphorus sediment loss was treated as an outflow and the area of the sediments was assumed equal to surface area, the mass balance equation could be expressed as:

$$V \frac{dP}{dt} = W - Q P - \nu A P \tag{13}$$

where ν was the apparent settling velocity of total phosphorus. This value was equal to $\alpha\nu'$ where α was the sinking velocity of settleable particulate phosphorus (SPP) and ν' was SPP/TP. If the steady state solution for P of equation 13 was substituted as above, R_p becomes:

$$R_p = \frac{\nu}{q_s + \nu} \tag{14}$$

Applying a least squares fit to the data from Kirchner and Dillon (1975) yielded an apparent settling velocity (ν) of 16 m/yr (Chapra 1975). Chapra's model assumed ν was constant among lakes. However, Reckhow (1979) subsequently showed that ν varied with q_s, specifically:

$$\nu = 11.6 + 0.2\, q_s \tag{15}$$

When compared to Kirchner and Dillon's results (equation 9), equation 14 with ν = 16 m/yr did not fit the actual data as well. Chapra (1975) explained that this in part may be due to lakes with lower areal water loads seem to exhibit a stronger relationship between R_p and q_s.

Dillon and Kirchner (1975) replied to Chapra's derivation of R_p by showing that equation 14 could be derived by multiplying the mean depth by the Dillon and

Rigler derivation (equation 4) or mathematically:

$$P = \frac{z_a \sigma}{z_a \sigma + z_a \rho} \tag{16}$$

which reduces to equation 14 where $z_a\rho = z_aQ/V = Q/A = q_s$ and $z_a\sigma = z_aK = \nu$ and hence:

$$R_P = \frac{\nu}{q_s + \nu} \tag{17}$$

The "1 -" of equation 4 is omitted because P sedimentation is now treated as "lost" instead of "retained".

Concurrently, Vollenweider (1975) found the approximate relationship:

$$\sigma = \frac{10}{z_a} \tag{18}$$

Substituting σ into the Dillon and Rigler derivation gives:

$$R_P = \frac{10}{10 + q_s} \tag{19}$$

which gives ν a value of 10 m/yr, compared to Chapra's 16. Dillon and Kirchner eliminated two outliers in the original data set and used a least squares fit to a homologous hyperbolic function and empirically derived:

$$R_p = \frac{13.2}{13.2 + q_s} \qquad (20)$$

which indicates $\nu = 13.2$ m/yr.

Larsen and Mercier's. Larsen and Mercier (1976) also examined factors affecting the phosphorus retention capacity of lakes. When oligotrophic and mesotrophic lakes which had P concentrations ≤ 25 μg/l comprised the data set, the best correlation with R_{exp} was found against $\ln q_s$ ($r = -0.92$). A least squares linear fit gave:

$$R_{exp} = 0.86 - 0.143 \ln q_s \qquad (21)$$

When two outlier lakes (Raven and Talbot) were eliminated, a correlation of -0.91 between R_{exp} and $\ln \rho_w$ resulted. The least squares linear fit yielded:

$$R_{exp} = 0.482 - 0.112 \ln \rho_w \qquad (22)$$

However, equations 21 and 22 predict R_p values greater than 1 for low q_s and ρ_w values (i.e., $q_s < 0.36$ m/yr or $\rho_w < 0.01$ yr^{-1}) and less than 0 for high q_s and ρ_w values (i.e., $q_s > 395$ m/yr or $\rho_w > 74$ yr^{-1}). In view of this anomaly, they regressed $\ln \sigma$ against $\ln \rho_w$ and derived:

$$\sigma = 0.761 \rho_w{}^{0\ 472} \qquad (23)$$

Substituting the result into:

$$R_P = \frac{\sigma}{\sigma + \rho_w} \qquad (24)$$

yielded:

$$R_P = \frac{1}{1 + 1.3\ \rho_w^{0.5}} \qquad (25)$$

Using a Gauss-Newton non-linear least squares algorithm resulted in:

$$R_P = \frac{1}{1 + 1.12\ \rho_w^{0.49}} \qquad (26)$$

Hence, Larsen and Mercier recommended using the following equation as a first approximation:

$$R_P = \frac{1}{1 + \rho_w^{0.5}} \qquad (27)$$

Canfield and Bachmann's. Jones and Bachmann (1978) showed that although the Vollenweider model worked well for natural lakes, it overestimated P values in several central Iowa artifical lakes by 3 - 10 times. The discrepancy appeared to be in under-estimating the phosphorus loss to the sediments (σ), because observed values in these lakes could be predicted if σ were increased by two orders of magnitude or greater. Canfield and Bachmann (1981) calculated σ from the steady state solution of the Vollenweider model to derive:

$$\sigma = \frac{L/z_a}{TP} - \rho \tag{28}$$

After constructing a correlation matrix with various data from 704 natural and artificial lakes, they empirically derived:

$$\sigma = 0.129 \left(\frac{L}{z_a}\right)^{0\ 549} \tag{29}$$

for both natural and artifical lakes,

$$\sigma = 0.162 \left(\frac{L}{z_a}\right)^{0\ 458} \tag{30}$$

for natural lakes, exclusively, and

$$\sigma = 0.114 \left(\frac{L}{z_a}\right)^{0\ 589} \tag{31}$$

for artifical lakes exclusively.

To accommodate a rapid initial sedimentation phase (presumably from suspended sediment adsorption), Canfield and Bachmann included an offset factor, f, of the loading term. Incorporating this factor and the sedimentation coefficient, they modify Vollenweider's original model to give the general form:

$$P = \frac{f\,L}{z_a\,(a(\frac{L}{z_a})^b + \rho)} \qquad (32)$$

By comparing predictions of this model with those of the models previously discussed, Canfield and Bachmann (1981) showed that their model had less predictive error and higher correlation coefficients against the observed values.

Reckhow's. As the above discussion indicates, many derivations of Vollenweider's model of phosphorus dynamics have been formulated and tested (Dillon 1974, Dillon and Rigler 1974b, Jones and Bachmann 1976, Larsen and Mercier 1976, Vollenweider 1976, Rast and Lee 1978, Canfield and Bachmann 1981, Reckhow 1988). Walker (1985) provides a comparison of many of these models that predict in-lake phosphorus concentrations. Reckhow (1988) generalized the continuity equation to describe any nutrient C as:

$$V(\frac{dC}{dt}) = M - Q\,C - k_C\,C\,V \qquad (33)$$

where C is in-lake nutrient concentration of C (mg/l), M is annual nutrient mass loading (g/yr), and k_C = nutrient trapping parameter specific to C (σ).

With the derivative set to zero (steady state) and dividing through by Q, solving for the in-lake nutrient concentration, C, gives:

$$C = \frac{C_1}{1 + k_C \tau_w} \tag{34}$$

Reckhow (1988) calibrated this model with cross-sectional data on southeastern lakes and reservoirs and derived the following equations as predictors of trophic state variables:

$$\log P = \log \frac{P_1}{1 + k_P \tau_w} \tag{35}$$

$$\log N = \log \frac{N_1}{1 + k_N \tau_w} \tag{36}$$

$$\log (\mathrm{chl})_{max} = 1.314 + \log (P^{0\ 321} \times N^{0\ 384} \times n_{CA}^{0\ 450} \times \tau_w^{0\ 136}) \tag{37}$$

$$\log SD = -0.470 + \log (P^{-0\ 364} \times \tau_w^{0\ 102} \times z_a^{0\ 137}) \tag{38}$$

where,

P = median summer total phosphorus concentration (mg/l),

P_1 = mean annual influent total phosphorus concentration (mg/l),

k_P = phosphorus nutrient trapping parameter (σ) (yr^{-1}),

N = median summer total nitrogen concentration (mg/l),

N_1 = mean annual influent total nitrogen concentration (mg/l),

k_N = nitrogen nutrient trapping parameter (yr^{-1}),
n_{CA}= number of chlorophyll samples, and
SD = Secchi disk depth (m).

Larsen and Mercier (1976) and Vollenweider (1976) related k_p with residence times and estimated k_p as $\tau_w^{-0\ 5}$. Canfield and Bachmann (1981) found k_p to be dependent on annual areal loading and τ_w (equation 29). By substituting L with P_1Q/A and z_a with V/A in equation 29 and simplifying Canfield and Bachmann's model can be reexpressed as:

$$\sigma = 0.129 \times P_1^{0\ 549} \times \tau_W^{-0\ 549} \tag{39}$$

Yet, Reckhow found regression parameters statistically different from zero for P_1, τ_w, and z_a, and thus derived the following model for calculating k_p using a nonlinear least squares method:

$$k_p = 3.0 \times P_1^{0\ 53} \times \tau_w^{-0\ 75} \times z_a^{0\ 58} \tag{40}$$

Reckhow (1988) also fitted the nitrogen trapping parameter, k_N, using the same method as that for k_p but found k_N to be dependent on τ_w, exclusively. This seems reasonable because nitrogen is more soluble and has lower particle adsorptive properties than phosphorus, which makes it less susceptible to inorganic sedimentary processes. Reckhow (1988) calculated k_N to be:

$$k_N = 0.67 \times \tau_w^{-0\ 75} \tag{41}$$

For statistics of the derivations of Reckhow's (1988) model, the reader is referred to the original paper.

Although Reckhow's model incorporates data from natural and artificial lakes for estimating these coefficients, it still operates under the CSTR assumption. Thornton et al. (1990) claims that this assumption is probably violated in reservoirs that exhibit strong longitudinal zonation.

Trophic State Indices

Carlson's (1977) trophic state index (TSI) provides a simple technique of trophic status evaluation based on the regression of three parameters, chlorophyll a (mg/m^3), total phosphorus ($\mu g/l$), and Secchi disk depth (m). This model assumes exclusive dependence among these parameters. The phytoplankton must be P-limited to insure chlorophyll a/total phosphorus dependence, albeit some phytoplanktonic species can exhibit luxuriant phosphorus uptake and thus bias the resulting trophic status evaluation (Wetzel 1983). Sakamoto (1966) found that chlorophyll a was proportional to P only when N:P was greater than ≈ 12.

Also, the algal contribution to transparency is assumed constant to insure the chlorophyll a, and therefore total phosphorus, dependence on Secchi disk depths. The intention was to provide a relatively simple method of trophic status classification that required the measurement of total phosphorus, chlorophyll a, or Secchi disk depth (Carlson 1977).

Carlson (1977) used a scale of TSI's from 0 to 100. A TSI less than ≈ 40 generally denotes oligotrophy, while a TSI above ≈ 55 denotes eutrophy. TSI's between ≈ 40 and 55 represent mesotrophy. He regressed the three parameters, set them equal to each other and derived the following computational forms of the TSI's:

$$\text{TSI (SD)} = 10 \times \left(6 - \frac{\ln \text{SD}}{\ln 2}\right) \tag{42}$$

$$\text{TSI (Chl)} = 10 \times \left(6 - \frac{2.04 - 0.68 \times \ln \text{Chl}}{\ln 2}\right) \tag{43}$$

$$\text{TSI (TP)} = 10 \times \left(6 - \frac{\ln \frac{48}{\text{TP}}}{\ln 2}\right) \tag{44}$$

Ideally, any value calculated from these equations should be equivalent regardless of the parameter used. Carlson (1977) warns, however, that use of this model only yields an index and thus does not define the trophic status. He further suggests that measurements be taken during the summer to avoid seasonal ambiguity (Carlson 1977). Megard et al. (1980) points out that biases in Carlson's Secchi disk index will exist in lakes with high non-algal turbidity. High non-algal turbidity also could bias the TSI(TP) by inducing light limitation.

Shannon and Brezonik (1972) formulated a similar index but based the TSI on primary productivity (PP, in mg C m^{-3} hr^{-1}), chlorophyll a (CA in mg m^{-3}), total organic nitrogen (TON, in mg/l), total phosphorus (TP, in mg P/l), Secchi disk transparency (SD, in m), specific conductance (COND, in μmho/cm), and a cation ratio ((Na+K)/(Ca+Mg)). The TSI is calculated as:

$$\begin{aligned} TSI = {} & 0.936(SD^{-1})+0.827(COND)+0.907(TON)+0.748(TP)+ \\ & 0.938(PP)+0.892(CA)+0.579(CR^{-1})+4.76 \end{aligned}$$

Values above ≈ 5 denote eutrophy, ≈ 1.2 to 5 denote mesotrophy, less than ≈ 1.2 denote oligotrophy (Reckhow and Chapra 1983). The major disadvantage of this model is the cost of obtaining the required parameters.

Porcella et al. (1980) formulated a TSI based in part on Carlson's model. This index, called the Lake Evaluation Index or LEI, requires morphometric data (for estimating macrophytic coverage), Secchi disk, total nitrogen, total phosphorus, chlorophyll a, dissolved oxygen profiles, and macrophytic development. The lack of bathymetric maps for many lakes and the extensive data required make this index less attractive. Porcella et al. (1980) recommends the LEI be used as a monitor to evaluate the effectiveness of restorative procedures (i.e., as an intra-lake index).

Reservoir Dynamics

Longitudinal Zonation

The bulk of allochthonous phosphorus and nitrogen loads to a reservoir typically originate in the drainage basin and enter the lake at its headwaters. Longitudinal processing in the reservoir dictate the fate of these nutrient loads (Kennedy et al. 1982). Thornton et al. (1981) and Kimmel and Groeger (1984) propose that this longitudinal processing occurs in three distinct zones - riverine, transition, and lacustrine.

The riverine zone occurs in the headwaters and maintains many of the river's characteristics such as the strong advective forces, high turbidity, and higher oxygenation, albeit organic carbon can contribute high oxygen demands (Thornton et al. 1990). Due to the high turbidity, the primary productivity (phytoplanktonic) in this zone is thought to be light-limited.

At the transition zone, the velocity slows which reduces advective forces. This reduction in advective forces allows finer silts and clays to settle and thus results in significant sedimentation. Nutrients, particularly phosphorus, will coprecipitate and be transported to the sediments due to adsorption to suspended sediment particles. Consequently, as the turbidity is reduced light penetration increases which when coupled with high levels of nutrients (i.e., N and P) induce a significant increase in primary productivity.

As the influent nutrients continue flowing towards the lower reaches, the lacustrine zone, dilution occurs making the nutrients less "bioavailable" and hence result in a decrease in primary productivity (Thornton et al. 1990).

The phytoplankton in this zone generally are thought to be nutrient-limited (usually P).

Trophic Status Evaluation

To define the trophic status of a lake means to classify its abiotic and biotic environment on the basis of nutrient content. The most commonly used classifications are oligotrophic (nutrient-poor), mesotrophic (intermediate), and eutrophic (nutrient-rich). Hypereutrophic, mesoeutrophic, etc. have been used to describe the same phenomena with greater sensitivity.

The traditional approach to trophic status evaluations has been directed towards estimating in-lake phosphorus concentrations and subsequently comparing these to the critical phosphorus levels proposed by Sawyer (1947), Vollenweider (1969), and Carlson (1977). This technique is valid if the phosphorus concentration reflects nutrient content and useful if it provides information on its impact on the biotic community, especially the phytoplanktonic productivity. The accurate prediction of in-lake phosphorus concentration and its inferred biotic impacts on productivity provide a means to predict future trophic states based on continued phosphorus loads to a system. This technique further provides a means by which the effects of external load reductions can be evaluated.

Predictive models of in-lake phosphorus concentrations have been used in reservoir assessment with limited success.

Toetz (1990) found that the natural lake model of Canfield and Bachmann (1981) most accurately predicted the in-lake phosphorus concentration of an Oklahoma reservoir, while the Reckhow model underestimated it. He further found that the Vollenweider (1969) model overestimated the phosphorus concentration. However, Toetz applied these models to an offset reservoir. This system would not be expected to have the plug flow characteristics of most mainstem impoundments, because it is artificially aerated and weakly stratifies, hence it approaches the homogeneous "natural lake" status (Toetz 1990). Given these characteristics, the high accuracy of the Canfield and Bachmann (1981) natural lake model and underestimation by Reckhow's (1988) model is not surprising.

This study by Toetz (1990), in effect, supports my contention. If a reservoir's longitudinal zonation can be identified, the researcher can omit data derived from the riverine and transition zone and only use the data from the lacustrine zone to assess the trophic status of the reservoir using the criteria developed for natural lakes, Carlson's, Vollenweider's, etc. This procedure would provide a more accurate predictive value from the available models. If the zones can not be identified, the lake of interest may be assumed to have enough similarity (i.e., homogeneity) to the lakes used in formulating the models and hence applicable in defining a reservoir's trophic status.

CHAPTER III

MATERIALS AND METHODS

Study Site

Grand Lake O' the Cherokees, located in Delaware, Mayes, and Ottawa counties, Oklahoma, was constructed in 1940 by the Grand River Dam Authority (GRDA). The impoundment provides flood control, hydroelectric power, and recreation. Grand Lake was formed by impounding the Grand River (called the Neosho River above the Spring River confluence). The distance from the confluence of the Neosho and Spring rivers to Pensacola Dam is approximately 69 km. The length of Grand Lake provides an ideal situation for studying longitudinal zonation.

The GRDA is currently conducting a water quality study in fulfillment of the requirements of a Federal Energy Regulatory Commission relicense. The water quality data obtained from their study and data from an EPA Phase I Clean Lakes study on Grand Lake conducted by the Water Quality Research Lab (WQRL) at Oklahoma State University was used in this study.

In order to complement the two studies, identical sampling stations were chosen. These stations included four mainstem stations and six accessory stations. The four mainstem stations were located about equidistant from one another extending from just below the Spring/Neosho confluence to just above Pensacola Dam (Fig. 3; for geographic coordinates see Table I). The six accessory stations were located about 3 km upstream from the Elk River confluence in the Elk River arm, about 1.5 km upstream from the Honey Creek confluence in the Honey Creek arm, at the Horse Creek confluence, Duck Creek confluence, Drowning Creek confluence, and in the west spillway directly below the Pensacola Dam.

Morphometry of Grand Lake

The morphometry of Grand Lake at normal pool elevation (Table II) was used in the Reckhow model for prediction of in-lake phosphorus and nitrogen concentrations, Secchi disk depths, chlorophyll a densities, probability of blue-green dominance, and probability of hypolimnetic anoxia.

Chemical Analyses

From May 1987 to October 1990, monthly profiles of dissolved oxygen, temperature, conductivity, and pH were recorded by GRDA using a Hydrolab model 4041 for all stations except the tailrace. Water samples also were collected and returned to the GRDA lab for subsequent analyses of alkalinity, Cl^-, SO_4^{2-}, total phosphorus, orthophosphate, total dissolved solids (TDS), turbidity (NTU), total hardness, K^+, Na^+, NH_3-N, NO_3-N, NO_2-N, total

TABLE I

GEOGRAPHIC COORDINATES OF GRAND LAKE SAMPLING SITES

Station ID	Latitude (North)	Longitude (West)
Honey Creek	36°33'50"	94°48'42"
Cowskin (Elk River)	36°38'59"	94°42'14"
Drowning Creek	36°30'42"	94°56'12"
Duck Creek	36°31'05"	94°58'06"
Horse Creek	36°37'30"	94°54'33"
Tailrace	36°27'30"	95°01'45"
Station #1	36°45'55"	94°47'05"
Station #2	36°38'25"	94°48'47"
Station #3	36°33'38"	94°54'42"
Station #4	36°29'42"	95°00'12"

Kjeldahl nitrogen (TKN), and total organic carbon (TOC).

I recorded temperature and dissolved oxygen profiles at the four mainstem stations on 07 Jun, 21 Jun, 21 Jul, 10 Aug, 07 Sep, and 02 Oct 89, and 10 Jul and 23 Aug 90. In addition to these profiles, I measured turbidity, conductivity, and pH of water samples collected with a 2-l van Dorn water sampler 0.5 m above the sediment and 0.5 m below the surface. Water samples were collected from these

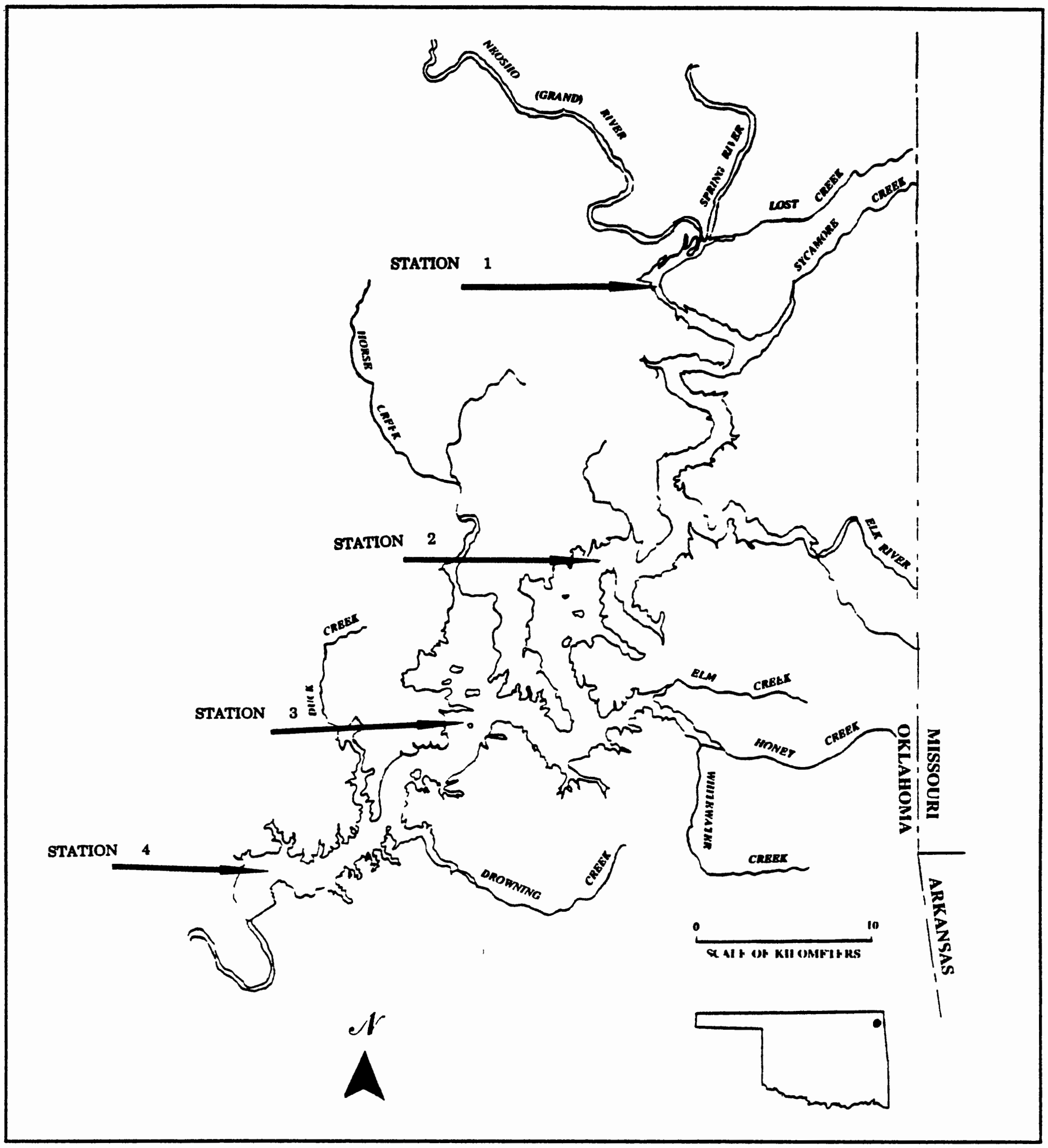

Figure 3. Grand Lake Mainstem Sampling Stations.

surface and bottom grabs in acid-rinsed high density polyethylene (HDPE) bottles, immediately placed on ice, and analyzed within 48 hr. These analyses included Na^+, K^+,

TABLE II

GRAND LAKE MORPHOMETRY AT NORMAL POWER POOL ELEVATION (OWRB 1990)

Parameter	Symbol	Value	
Length of pool	l_p	69	km
Shoreline length	L	1005	km
Surface area	A_o	1.88×10^4	ha
Volume	V_o	2.05×10^9	m^3
Mean depth	z_a	10.9	m
Maximum depth	z_m	36.6	m
Shoreline development	D_L	20.7	
Hydraulic retention time	τ_w	0.36	yr

NH_4^+, Mg^{2+}, and Ca^{2+} using a Waters HPLC and Cl^-, *o*-P, NO_2-N, NO_3-N, and SO_4^{2-} using a Dionex System 12 Ion Chromatograph (IC). All samples except total phosphorus aliquots were filtered prior to analysis through a 0.45 μm filter. Total phosphorus was analyzed using methods described by Hach Water Analysis Handbook, which is equivalent to the persulfate digestion/molybdate colorimetric procedure (Lind 1985). I also collected samples 0.5 m below the surface by the techniques previously described but decanted to 250-ml opaque non-acid-washed HDPE bottles, immediately placed on ice, and returned to the WQRL for subsequent chlorophyll a analysis using methods described by the American Public Health Association (APHA 1981). Triplicate samples of randomly selected stations (1 per trip), field blanks, EPA QA-QC aliquots, and spiked samples were used for quality assurance/control.

Statistical Analyses

Longitudinal zonation of physical, chemical, and biological data were statistically tested for differences between the four mainstem stations using the Mann-Whitney test as described by Gilbert (1987). The statistical software package WQSTAT was used for performing these tests (Phillips et al. 1988). Data sets that were too small for use with WQSTAT II were tested as per the Mann-Whitney procedure using a customized spreadsheet in Quattro Pro. Distribution-free tests, although valid, usually do not have the power (i.e., 1 - ß) that distribution-dependent tests denote (Steel and Torrie 1980). The Mann-Whitney test was chosen because it has been shown to be about 95% as powerful as the two-sample _t_-test when the normality assumptions are not violated; when these are violated the Mann-Whitney test is many times more powerful (Mood 1954, Hodges and Lehman 1956, Zar 1984).

The parameters tested and used as criteria for longitudinal zonation were total phosphorus, TOC, TKN, turbidity, and chlorophyll a because they reflect biotic activity, such as phytoplanktonic productivity, and abiotic activity, such as coprecipitation and suspended sediment adsorption and/or sedimentation.

Trophic Status Evaluation

When significant differences among stations in Grand Lake were detected, further evidence of Thornton's (1982) proposed zonation was investigated by evaluating reduced turbidity coupled with increased productivity in the transition zone. Nutrient and chlorophyll data from a WQRL extensive survey conducted on 10 Jul 90 were used to assess these zones of different productivity.

The trophic status of Grand Lake was evaluated on all data and only lacustrine zone data using Carlson's criteria. The rationale for this approach is that the lacustrine zone more closely approximates natural lake conditions and thus would provide a more appropriate comparison to trophic state indices formulated from data on natural lakes. The data on the most downstream station, which is the most likely to exhibit lacustrine characteristics, could have been used exclusively. However, I believe that if more upstream stations also exhibit lacustrine characteristics, the trophic status of the lake can be more accurately assessed with these data included.

The resulting classifications were compared for congruency. Theoretically, Carlson's TSI(Chl), TSI(TP), and TSI(SD) should be equivalent if the assumptions are met. The differences (residuals) among the TSI's were used as indices of agreement. Reduced residuals (i.e., better agreement among the TSI's) should denote a more accurate classification of the lake.

Restoration Potential

Reckhow's model was programmed in BASICA and executed using reduced influent phosphorus and nitrogen concentrations. The resulting predictions of P, N, Secchi disk depths, maximum chlorophyll a, probability of blue-green dominance, and probability of an anoxic hypolimnion were evaluated in terms of trophic status. The percent reduction of external phosphorus and nitrogen loads which yield oligotrophic and mesotrophic conditions were estimated based on the model's predictions.

CHAPTER IV

RESULTS AND DISCUSSION

Descriptive Statistics

Grand Lake showed a decreasing longitudinal trend in mean values of TOC, TKN, turbidity, and chlorophyll a from station 1 to station 4 (Table III). The same trend existed for mean values of TP except station 4 mean TP was slightly higher than station 3. All mean values of TP indicate eutrophic conditions as per Sawyer's (1947) criteria. Mean chlorophyll a values for stations 1 and 2 indicate eutrophic conditions, while stations 3 and 4 indicate mesoeutrophy (Table IV).

Longitudinal Zonation

Total Phosphorus

Station 1 TP was significantly higher ($\alpha = 0.01$) than the other three mainstem stations (Table V). Station 2 TP also was significantly higher ($\alpha = 0.01$) than stations 3 and 4. Station 3 TP was not significantly different from station 4.

If statistically significant differences of TP among the stations are used as criteria for zonation, the riverine and transition zones occur above station 3. The precise

TABLE III

STATISTICS OF PHYSICOCHEMICAL DATA[1] USED FOR TESTING LONGITUDINAL ZONATION IN GRAND LAKE

PARAMETER	STATION	MEAN	MEDIAN	s	N
TOTAL	1	0.180	0.113	0.181	29
PHOSPHORUS	2	0.171	0.070	0.348	37
(mg/l)	3	0.034	0.052	0.141	37
	4	0.054	0.017	0.074	36
	DROWNING	0.118	0.036	0.265	34
	DUCK	0.049	0.031	0.038	35
	ELK	0.107	0.044	0.119	16
	HONEY	0.105	0.037	0.179	36
	HORSE	0.085	0.042	0.162	36
	TAILRACE	0.059	0.034	0.037	20
TOTAL	1	9.757	7.125	5.051	30
ORGANIC	2	9.629	6.745	6.169	37
CARBON	3	7.921	6.350	2.980	37
(mg/l)	4	7.654	6.490	2.896	37
	DROWNING	8.310	6.445	4.756	37
	DUCK	8.865	7.440	5.356	37
	ELK	10.235	4.650	5.781	17
	HONEY	8.161	6.065	5.059	37
	HORSE	8.824	6.705	5.508	37
TOTAL	1	18.252	1.715	18.574	29
KJELDAHL	2	14.601	1.855	15.069	37
NITROGEN	3	11.545	1.875	10.757	37
(mg/l)	4	9.534	1.125	9.472	37
	DROWNING	7.640	1.175	7.919	37
	DUCK	8.854	1.450	8.441	37
	ELK	16.487	11.608	8.006	15
	HONEY	10.787	1.325	11.976	37
	HORSE	11.228	2.150	11.773	37
TURBIDITY	1	29.507	16.500	32.449	30
(NTU)	2	15.324	8.350	15.656	37
	3	7.964	5.150	7.342	37
	4	6.377	3.000	7.456	37
	DROWNING	6.227	3.000	7.123	37
	DUCK	5.869	3.200	6.064	37
	ELK	11.938	3.600	27.940	17
	HONEY	7.722	4.500	7.694	37
	HORSE	8.274	6.000	6.860	37
	TAILRACE	5.550	3.000	6.207	20

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[1] Data from GRDA.

TABLE IV

STATISTICS OF CHLOROPHYLL a DATA[1] ($\mu g/l$) USED FOR TESTING LONGITUDINAL ZONATION IN GRAND LAKE

STATION	MEAN	MEDIAN	s	N
1	25.2	26.0	18.1	8
2	17.6	13.8	16.7	8
3	9.5	8.6	5.2	7
4	8.4	8.9	7.2	6

[1] Data from WQRL.

location of the riverine/transition border cannot be identified exclusively on these results because significant differences were found between stations 1 and 2, stations 1 and 3, and stations 2 and 3. However, these data seem to suggest that the riverine zone includes station 1, transition zone includes station 2, and stations 3 and 4 are within the lacustrine zone.

Total Organic Carbon

No significant difference between stations 1 and 2 was detected (Table V). However, significant differences (α = 0.05) between stations 1 and 3 (H_A: $\mu_1 > \mu_3$) and stations 1 and 4 (H_A: $\mu_1 > \mu_4$) were detected. The remaining possible comparisons (among stations) showed no significant differences.

Using TOC as a criterion for zonation, the riverine zone occurs above station 2. Designating station 1 as part of this zone implies the existence of the transition zone below station 2; the data clearly do not support such a statement. Therefore, given the significant differences found in TOC, only two zones can be inferred with their boundary occurring above station 2, albeit which two (assuming all three exist) cannot be ascertained based exclusively on these TOC data. It is possible, however, that all three zones exist and are identifiable from TOC data but the stations chosen in this study were not allotted in all three zones and hence not detected.

Turbidity

Station 1 turbidity was found to be significantly higher ($\alpha = 0.01$) than that of stations 2, 3, and 4. Station 2 turbidity also was significantly higher ($\alpha = 0.01$) than at stations 3 and 4 (Table V). Turbidity at station 3 was significantly higher ($\alpha = 0.05$) than station 4.

These data suggest a trend towards homogeneity in turbidity as station 4 is approached. Based on these relationships, station 1 lies in the riverine zone, station 2 in the transition zone, and stations 3 and 4 in the lacustrine zone. This arrangement also is supported by the rapid drop in turbidity, characteristic of the transition zone, at the Elk River confluence.

Total Kjeldahl Nitrogen

Although the tests of station 1 against station 2 and

TABLE V

STATISTICAL COMPARISONS USING THE MANN-WHITNEY TEST OF VARIABLES AT THE FOUR STATIONS IN GRAND LAKE

		STATION		
	STATION	2	3	4
Total Phosphorus	1	**	**	**
	2	--	**	**
	3	--	--	ns
Total Organic Carbon	1	ns	*	*
	2	--	ns	ns
	3	--	--	ns
Turbidity	1	**	**	**
	2	--	**	**
	3	--	--	*
Total Kjeldahl Nitrogen	1	ns	*	**
	2	--	ns	*
	3	--	--	ns
Chlorophyll a	1	ns	*	ns
	2	--	ns	ns
	3	--	--	ns
Chlorophyll a (excluding 21 Jul 89)	1	ns	*	*
	2	--	ns	ns
	3	--	--	ns

* = significant ($\alpha = 0.05$).
** = highly significant ($\alpha = 0.01$).
ns = not significant.

stations 2 against 3 showed no significant differences in TKN, station 1 TKN was significantly higher than station 3 TKN (Table V). The difference between stations 1 TKN and 4 TKN was highly significant ($\alpha = 0.01$). Station 2 TKN also was significantly higher ($\alpha = 0.05$) than station 4 TKN. Station 3 TKN was not significantly different from station 4 TKN.

These results suggest that weak changes in zones occur near station 3. Also, a change in zones is indicated between stations 2 and 4 (i.e., station 2 significantly different than station 4). These results do not agree with those of the TP and turbidity.

Chlorophyll a

With all chlorophyll a data from the mainstem stations included in the analysis, the only significant difference detected was station 1 against station 3 (H_A: $\mu_1 > \mu_3$; $\alpha = 0.05$) (Table V). If the proposed zonation exists, station 1 should have been significantly higher than station 4. Two possible explanations exist. First, the higher urban development at the lower end of the lake (i.e., station 4 region) is contributing higher nutrient loads from sewage effluents, runoff, etc. This could be increasing primary productivity and thus elevating station 4 chlorophyll to values statistically similar to those of stations 1 and 2. Mean TP at station 4 (54 μg P/l) was higher than station 3 (34 μg P/l), but the difference was not statistically significant (Tables III and V). This explanation is still plausible, however, because biotic responses may not always adhere to the rules of statistical inference.

It is possible that data outliers are affecting the tests. These two explanations are not mutually exclusive. Non-parametric tests are supposedly less sensitive to outliers. However, the effect of outliers becomes more pronounced when the data sets are small, as is this chlorophyll data. If this is affecting the outcomes of the tests, the most obvious outlier data are those measured on 21 Jul 89. With these data eliminated, the same tests resulted in signficant differences between stations 1 and 3 and stations 1 and 4 (H_A: $\mu_1 > \mu_3$ and H_A: $\mu_1 > \mu_4$; $\alpha = 0.05$) (Table V). The remaining possible comparisons were not significant. These results (Table V) suggest a change in productivity between stations 1 and 3, and therefore a change in zonation is indicated.

The chlorophyll and turbidity data from an extensive survey conducted by personnel of WQRL on 10 Jul 90 reveal the most conclusive evidence that suggests the location of the transition zone (Fig. 4). These data clearly show a marked decrease in turbidity and a concurrent increase in productivity at the Elk River confluence. The Elk River might be contributing higher nutrients and hence, causing the increased productivity. However, this could not be shown statistically. Ostensibly, the transition zone begins slightly north of the Elk River confluence and gradually merges with the lacustrine zone between stations 2 and 3.

Conclusions and Discussion

The parameters used in this study as criteria for

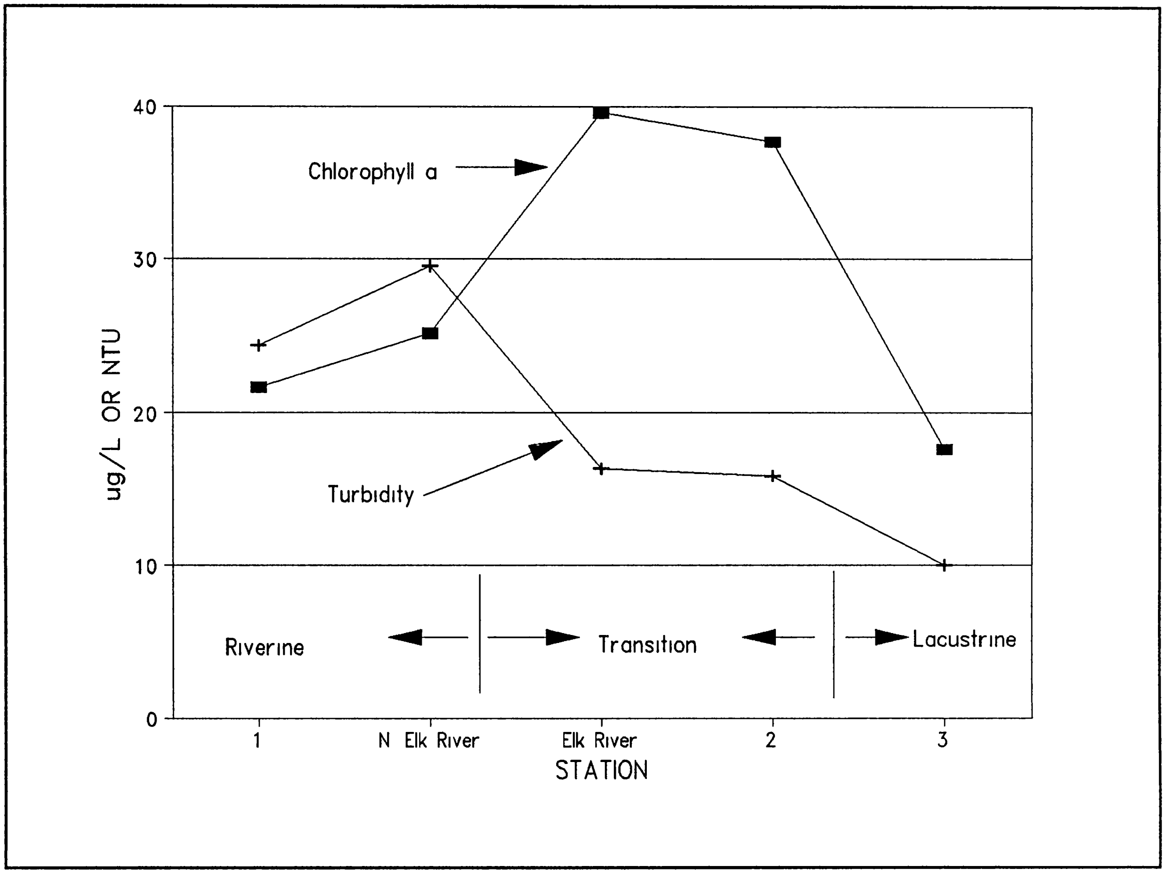

Figure 4. Longitudinal Variation in Chlorophyll a and Turbidity at the Upper Three Stations in Grand Lake on 10 Jul 90.

longitudinal zonation suggest that identifiable zones as described by Thornton et al. (1981) and Kimmel and Groeger (1984) exist in Grand Lake. Riverine characteristics are exhibited at station 1. These characteristics gradually give way to transitional characteristics north of the Elk River confluence and lacustrine characteristics between stations 2 and 3. Therefore, stations 3 and 4 are designated as the lacustrine zone.

All statistical tests were not in agreement. Tests of TP and turbidity (Table V) showed similar results. However, tests for TKN and TOC did not reflect zonation as strongly. Different results for TKN might be due to greater solubility of nitrogenous compounds than phosphorus compounds. Hence, significant differences would not be detected until the effects of dilution are pronounced (no two adjacent stations showed significance). The reasons for different results in TOC remains unexplained.

These differences may be explained by the findings of Vannote et al. (1980) which states that biotic communities in rivers (and hence biotic dynamics) occur along a continuum. The same phenomena may occur in highly serpentine reservoirs, e.g., Grand Lake, which can exhibit characteristics of long slow-flowing rivers, especially in the upper reaches.

Trophic Status Evaluation

Before Correction

Carlson's (1977) trophic state index (TSI) calculated from averages of all four mainstem stations indicated eutrophic conditions (i.e., $TSI > \approx 55$) on all sampling dates except the TSI(Chl) derived on 06 Jun 89 data which indicated mesotrophic conditions (i.e., $40 \leq TSI \leq 55$) (Fig. 5). TSI(Chl) on 06 Jun 90 indicated meso-eutrophic conditions. TSI(TP) was calculated to be greater than TSI(Chl) and TSI(SD) which were approximately the same.

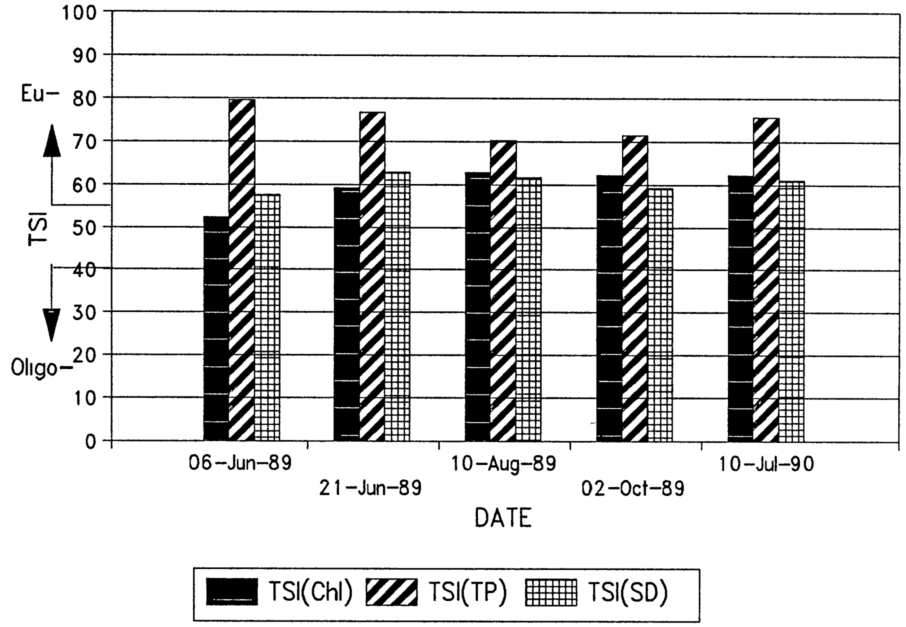

Figure 5. Carlson's TSI's Calculated from Averages of All Four Mainstem Stations on Grand Lake (Data from WQRL).

This pattern of error is consistent on all dates sampled (Fig. 5).

After Correction

To correct for longitudinal zonation, station 1 and stations 1 and 2 were deleted from the data set. TSI's were recalculated and tested for congruency. The TSI's calculated from data excluding only station 1 indicated

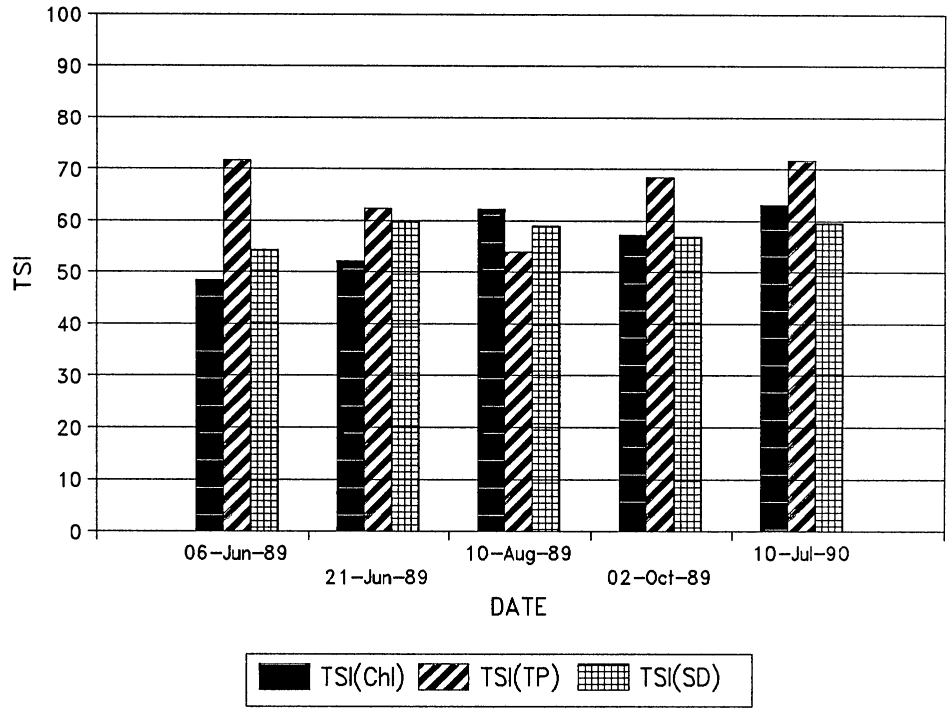

Figure 6. Carlson's TSI's Calculated from Averages of Mainstem Stations 2, 3, and 4 (Data from WQRL).

meso-eutrophic conditions (Fig. 6). The TSI's calculated from data excluding stations 1 and 2 (i.e., data from lacustrine zone) also indicated meso-eutrophic conditions (Fig. 7).

TSI Errors

The relative magnitude of the difference in TSI's provides an estimate of the residual (i.e., error) in trophic status classification. Residual is used here as the

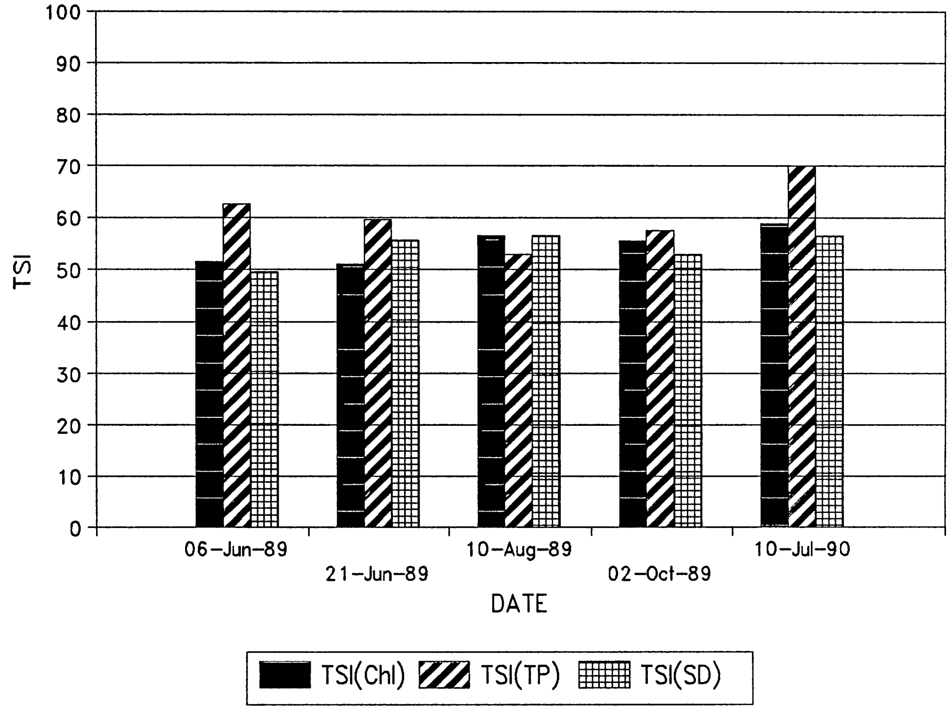

Figure 7. Carlson's TSI's Calculated from Averages of Mainstem Stations 3 and 4 (Data from WQRL).

absolute value of [TSI(X) - TSI(Y)], where X and Y are TP, Chl, or SD. As stated earlier, the three TSI's ideally should be calculated as the same (i.e., residual = 0). Therefore, the residual can be used as an index of agreement among the TSI's for a given date (or station); a smaller residual reflects more agreement.

Correction for zonation yielded the least TSI(Chl)/TSI(TP) residual when data from stations 1 and 2

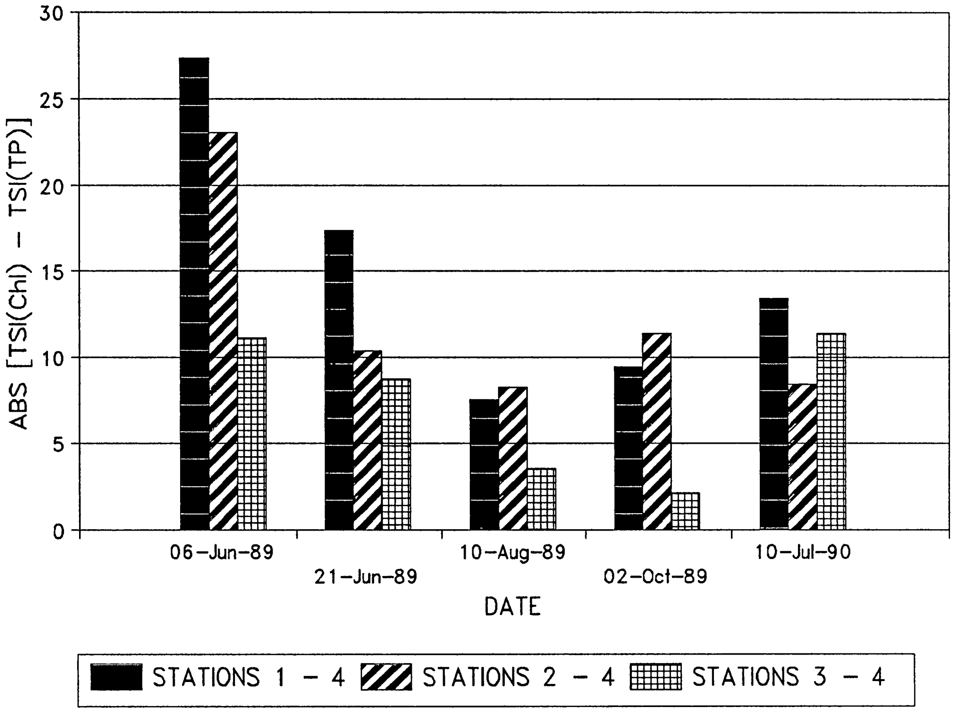

Figure 8. Residuals of Carlson's TSI(Chl)/TSI(TP) Before and After Correction for Longitudinal Zonation.

were deleted on all dates except 10 Jul 90, which had the least residual with only station 1 eliminated (Fig. 8). The residual with stations 1 and 2 eliminated still was less than with all four stations included.

These results agree with the zonation concept, which postulates that phytoplankton in the lacustrine zone are nutrient-limited. Therefore, the good agreement (low residual) between TSI(Chl) and TSI(TP) when only lacustrine data are used is readily explainable, albeit limiting factors were not tested.

The TSI(Chl)/TSI(SD) residuals of stations 3 and 4, exclusively, were consistently less than that for stations 2, 3, and 4, excluding the data from 02 Oct 89 (Fig. 9). This discrepancy on 02 Oct 89 may be related to autumn mixing, while turbidity (indirectly expressed as SD) is somewhat homogeneous, especially in the transition and lacustrine zones.

Including all four stations in calculating the TSI's yielded smaller residuals than those with only the riverine zone (station 1) excluded (Fig. 9). This probably is due to the association of high turbidities and high chlorophyll densities at station 1 overshadowing the relative residuals of the other stations.

The least TSI(TP)/TSI(SD) residuals resulted when the the riverine and transition zone data were excluded on all dates except 21 Jun 89 and 10 Jul 90 (Fig. 10). In both of these exceptions, eliminating only the riverine zone yielded the smallest residual. Also in both cases, the residuals were only slightly smaller than residuals resulting from excluding riverine and transition zone data. In all cases, correcting for zonation, whether eliminating only riverine data or riverine and transition zone data, improved the agreement between TSI(TP) and TSI(SD) on all dates.

Correlation matrices (Table IV) were constructed before and after correction for zonation. These matrices were based on data that did not include 20 Jul 90 data because it

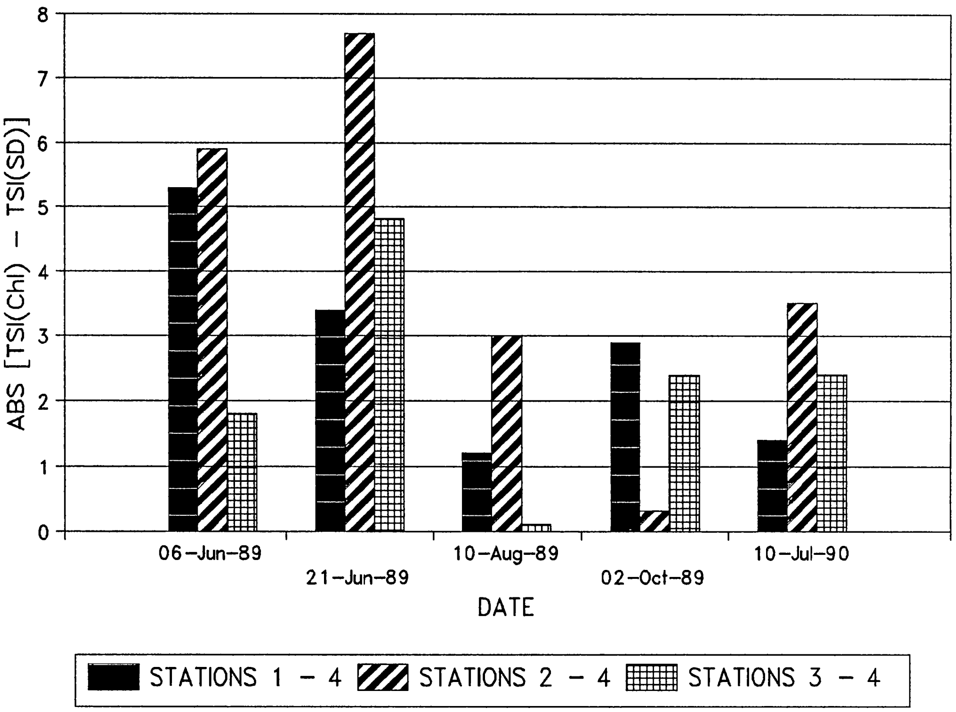

Figure 9. Residuals of Carlson's TSI(Chl)/TSI(SD) Before and After Correction for Longitudinal Zonation.

was designated as an outlier and 23 Aug 90 data because the TP data for station 4 was below detectable quantities and hence does not provide a comparison. The correlation coefficients (r) decreased after correction for zonation in every case except TSI(Chl) against TSI(SD). The improvement in TSI(Chl) against TSI(SD) probably is due to the decreased inorganic turbidity and subsequent increase in the effect of algae on turbidity. However, these correlation-based results have an inherent bias. Smaller numbers (i.e.,

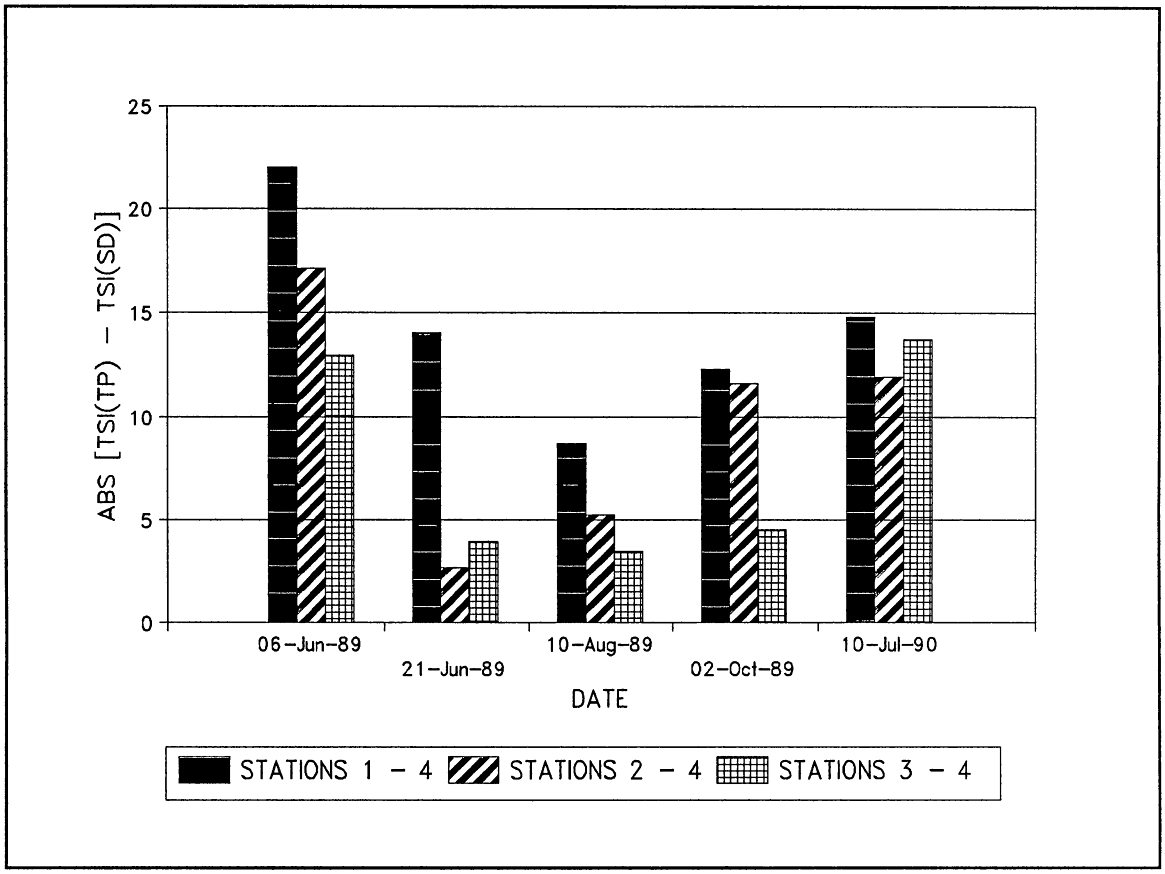

Figure 10. Residuals of Carlson's TSI(TP)/TSI(SD) Before and After Correction for Longitudinal Zonation.

TSI's), which are characteristic of the lacustrine zone relative to the riverine and transition zones, with the same residuals will have a smaller correlation. Essentially, the range is decreased while the residual remains the same. Therefore, the same residual will reflect a smaller correlation and hence imply more residual when it doesn't exist.

TABLE VI

CORRELATION COEFFICIENTS BETWEEN TSI'S AT FOUR STATIONS IN GRAND LAKE

Stations	Variables	TSI(TP)	TSI(SD)
1,2,3,4	TSI(Chl)	0.88	0.30
	TSI(TP)	--	0.23
2,3,4	TSI(Chl)	0.43	0.16
	TSI(TP)	--	0.16
3,4	TSI(Chl)	0.24	0.54
	TSI(TP)	--	0.03

Restoration Potential

The following results of Reckhow's (1988) model were calculated using the program listed in Appendix A. These results are theoretical predictions derived from a conceptual model that was empirically fitted to existing lake/reservoir data. Consequently, the data may not reflect the actual occurrences, especially where nutrient reductions yield predicted in-lake nutrient concentrations that are outside the range of the calibration data set. However, this model is chosen because it has a sound conceptual basis and was empirically fitted to actual reservoir data.

Nitrogen Reductions

Assuming phosphorus inputs were unchanged, the model predicts that mesotrophic conditions at station 4 could be attained if total N-inputs were reduced by ≈ 58%, using Sawyer's (1947) criterion that mesotrophy is 300 μg N/l $\leq$ in-lake N $\leq$ 600 μg N/l (Fig. 11). The resulting chlorophyll density is predicted to be ≈ 11.2 μg/l (Fig. 11). The resulting probabilities of an anoxic hypolimnion and blue-green algal dominance were predicted to be ≈ 0.77 and ≈ 0.84, respectively (Figs. 12 and 13). The high probability of blue-green dominance would be expected because reducing N gives nitrogen-fixers (e.g., blue-greens) a competitive advantage. Hence, the inverse relationship in Figure 13. The changes in Secchi disk depths to reduced N-inputs were predicted to be neglible (Fig. 14).

Again assuming P-inputs remain constant, the model predicted that oligotrophic conditions as per Sawyer (1947) at station 4 would require ≈ 80% reduction of N-inputs (Fig. 11). The resulting chlorophyll a density with this reduction was predicted to be ≈ 8.8 μg/l (Fig. 11). The probabilities of hypolimnetic anoxia and blue-green dominance were predicted as ≈ 0.73 and 0.91, respectively (Figs. 12 and 13).

Phosphorus Reductions

Assuming N-inputs remain constant, mesotrophic conditions (i.e., TP $\leq$ 20 μg P/l) at station 4 would require ≈ 60% reduction of TP-inputs as predicted by the model (Fig. 15). The resulting chlorophyll a density was predicted to be ≈ 13.5 μg/l (Fig. 15). The probabilities of hypolimnetic anoxia and blue-green dominance were predicted as ≈ 0.64 and ≈ 0.41, respectively (Figs. 16 and 17). The predicted Secchi disk with 60% reduction of P-input is ≈ 1.78 m (Fig.

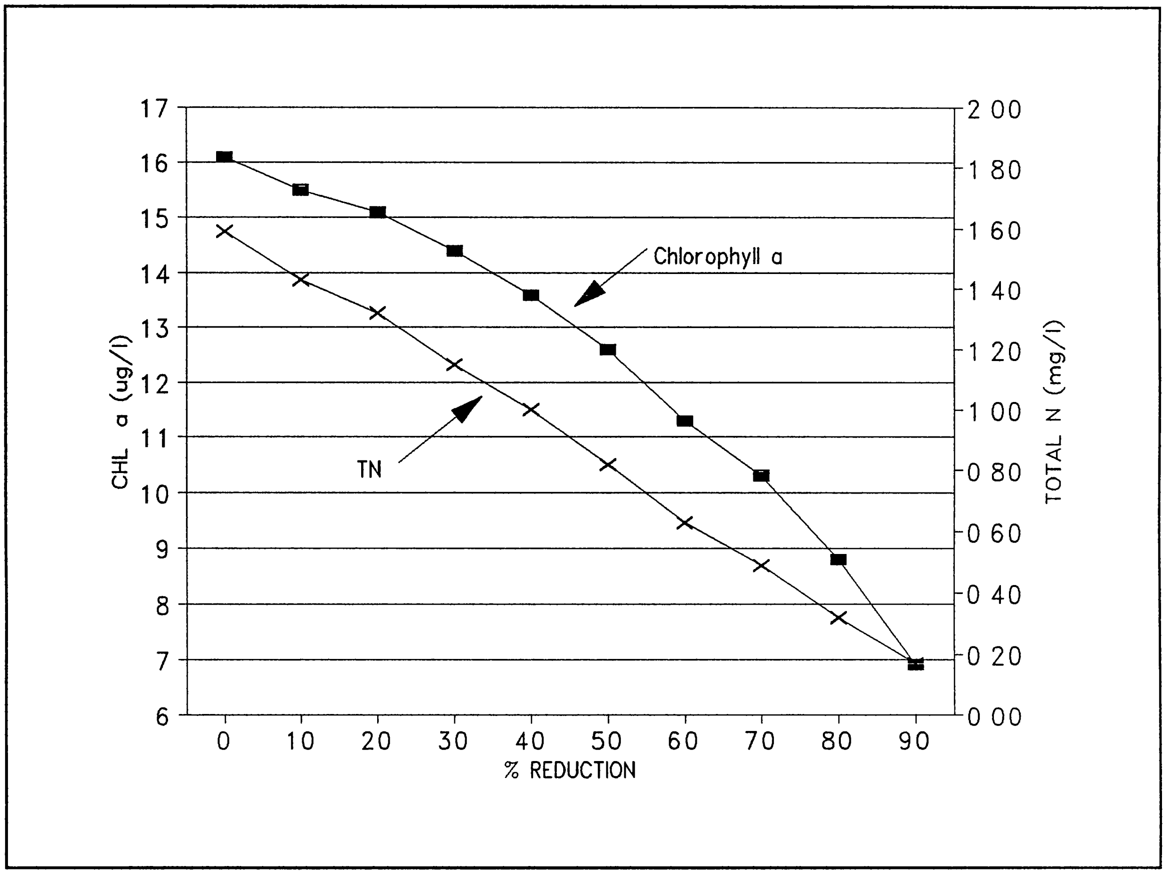

Figure 11. Predicted Chlorophyll a and TN at Station 4 vs % Reduction of N-inputs.

18).

Oligotrophic conditions (i.e., TP ≤ 10 μg/l) at station 4 would require ≈ 85% reduction of P-inputs, assuming N-inputs remain constant (Fig. 15). The resulting chlorophyll a density was predicted to be ≈ 10.8 μg/l (Fig. 15). The probabilities of hypolimnetic anoxia and blue-green dominance with ≈ 85% reduction of P-inputs were predicted to be ≈ 0.37 and ≈ 0.18, respectively (Figs. 16 and 17). Secchi disk depth at this reduction is predicted as ≈ 2.3 m

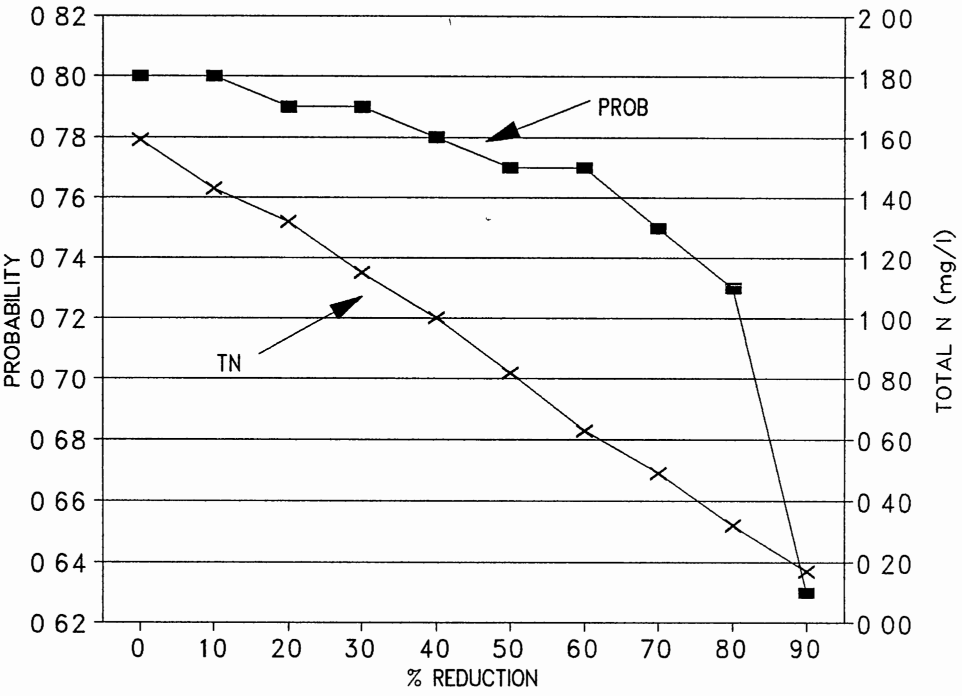

Figure 12. Predicted Probability of Hypolimnetic Anoxia and TN at Station 4 vs % Reduction of N-inputs.

(Fig. 18).

Conclusions and Discussion

Reckhow's (1988) model predicts that maximum benefit, in terms of trophic state variables, would be achieved by phosphorus reduction. The Secchi disk depths and probabilities of hypolimnetic anoxia and blue-green dominance were predicted to be more sensitive to P-reduction

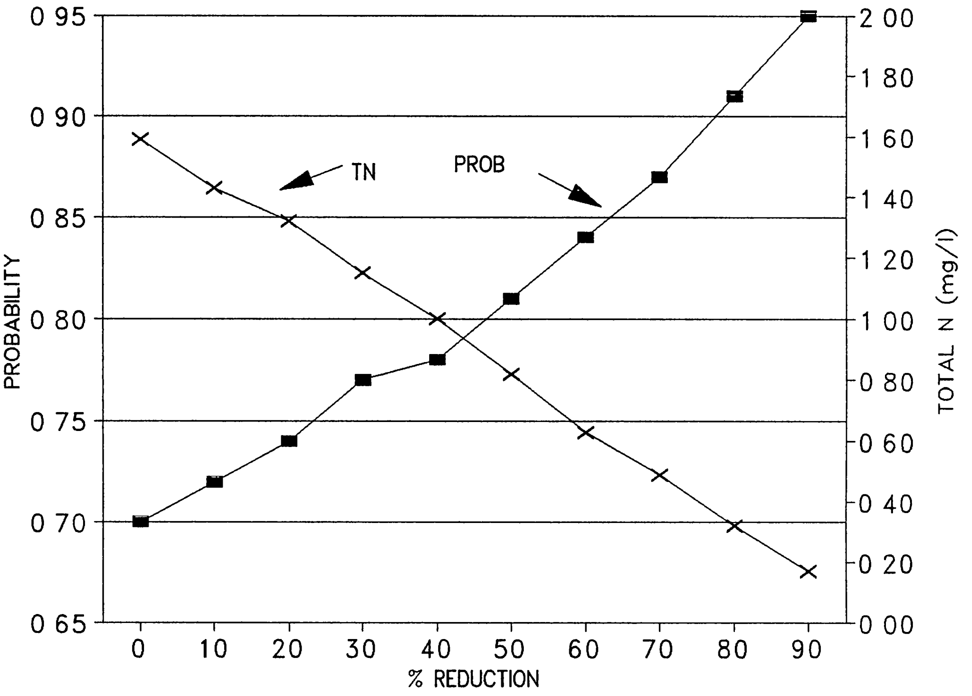

Figure 13. Predicted Probability of Blue-green Dominance and TN at Station 4 vs % Reduction of N-inputs.

than N-reduction, while chlorophyll a was predicted to be slightly more sensitive to N-reductions.

Based on these predictions and the threat of sediment release of toxic metals, P-reduction measures should be implemented throughout the basin. This approach is aimed primarily at reducing the probability of anoxia which increases an already significant threat to toxic metal release. This may be accomplished by two methods; 1) voluntary ban on phosphate-based detergents and 2) public

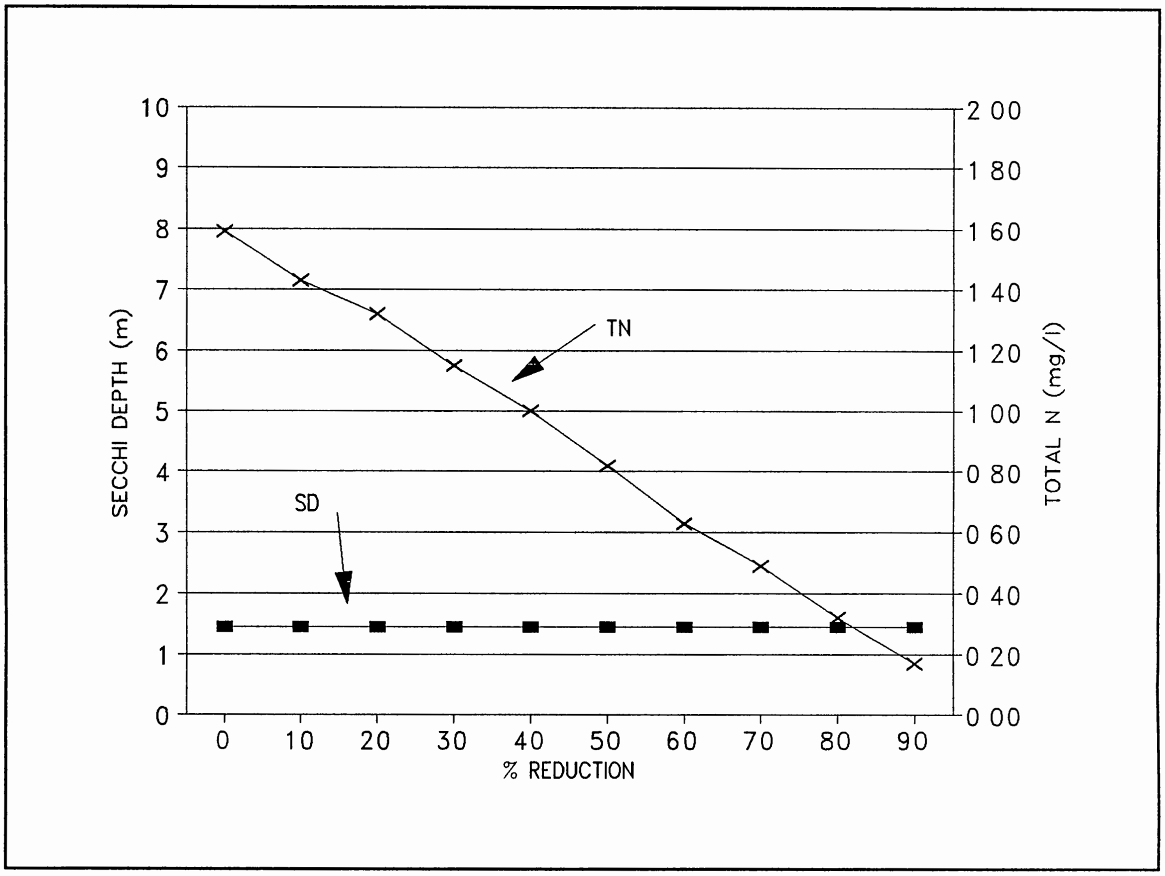

Figure 14. Predicted Secchi Depths and TN at Station 4 vs % Reduction of N-inputs.

awareness programs on BMPs (best management practices) throughout the drainage basin.

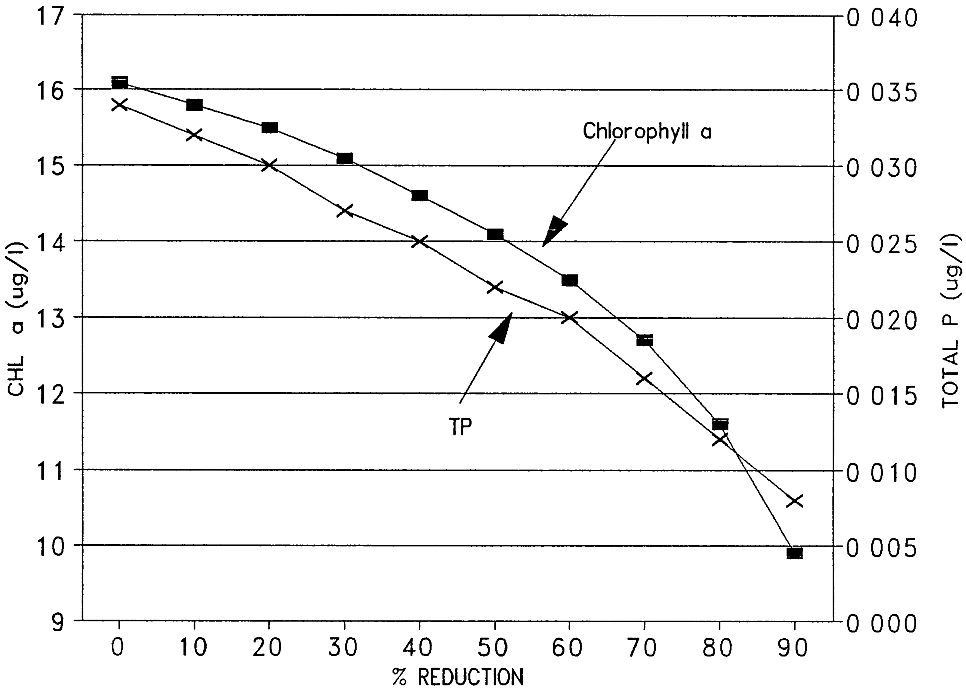

Figure 15. Predicted Chlorophyll a and TP at Station 4 vs % Reduction of P-inputs.

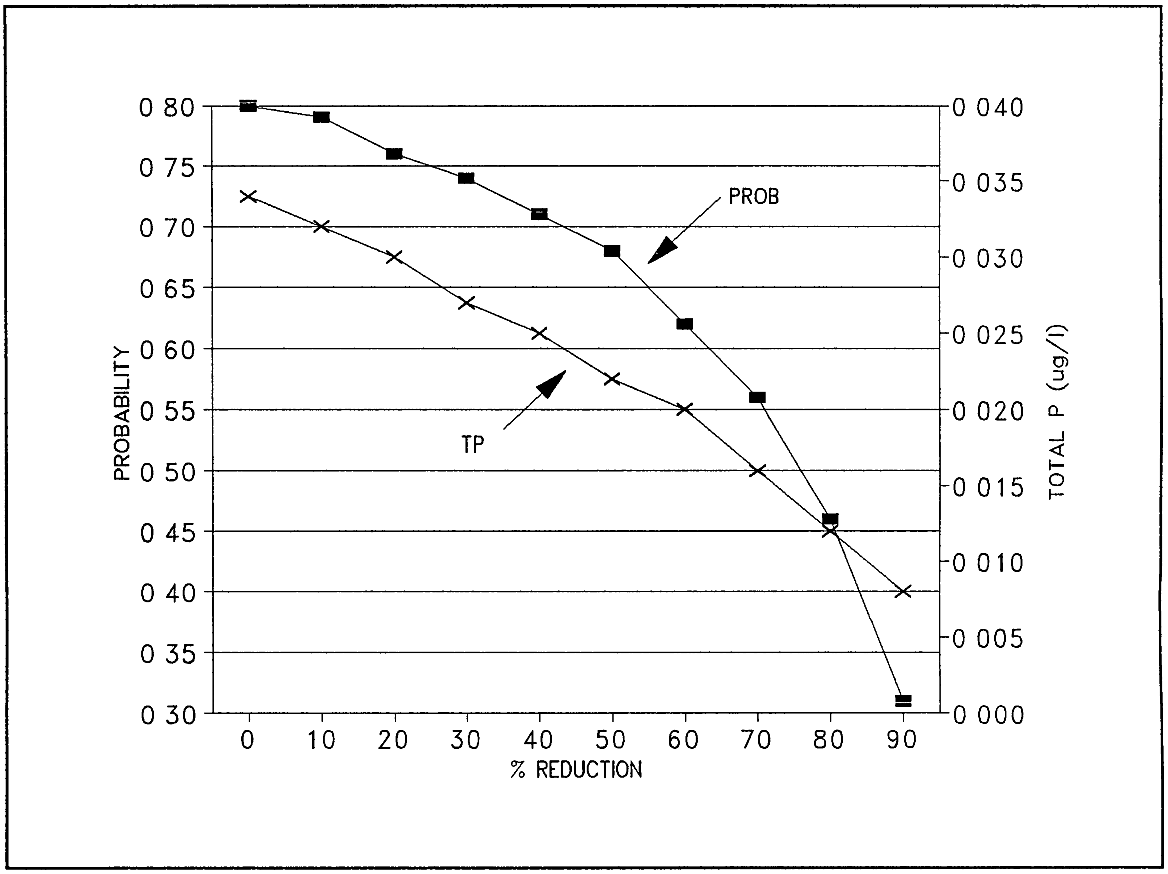

Figure 16. Predicted Probability of Hypolimnetic Anoxia and TP at Station 4 vs % Reduction of P-inputs.

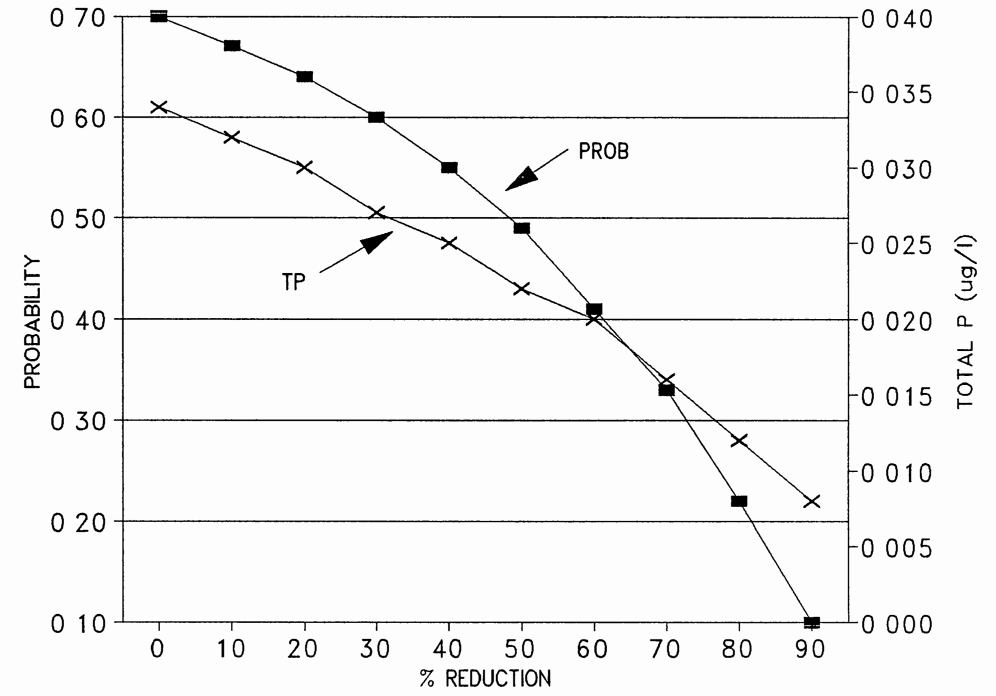

Figure 17. Predicted Probability of Blue-green Dominance and TP at Station 4 vs % Reduction of P-inputs.

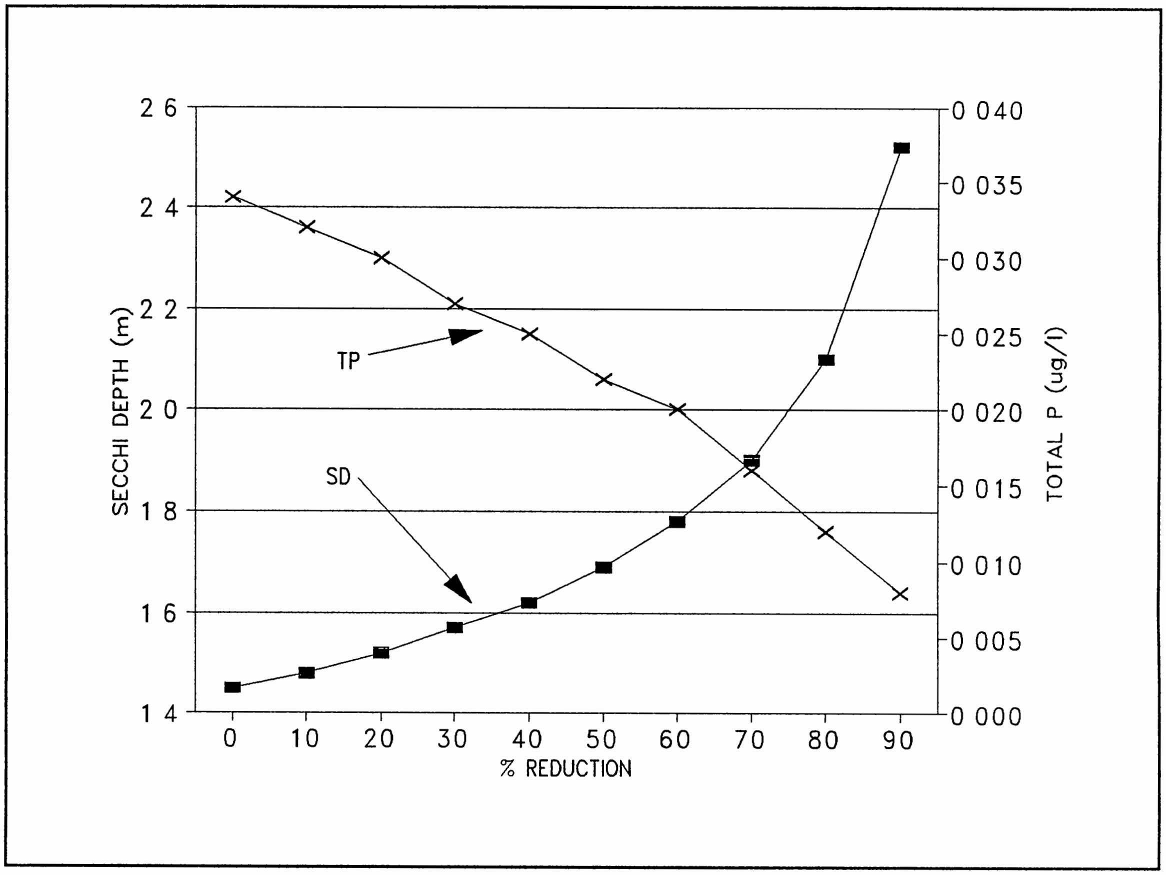

Figure 18. Predicted Secchi Depths and TP at Station 4 vs % Reduction of P-inputs.

LITERATURE CITED

American Public Health Association. 1975. Standard Methods for Examination of Water and Wastewater. 14th ed. American Public Health Association, Washington D. C., 1193 pp.

Bachmann, R. W. and J. R. Jones. 1974. Phosphorus inputs and algal blooms in lakes. Iowa State J. Res., Ames IA, 49:155-160.

Bandurski, R. S. 1965. Biological reduction of sulfate and nitrate. In J. Bonner and J. Varner, eds. Plant Biochemistry. Academic Press, New York NY, pp. 467-490.

Burris, R. H., F. J. Eppling, H. B. Wahlin, and P. W. Wilson. 1943. Detection of nitrogen-fixation with isotopic nitrogen. J. Biol. Chem. 148:349-357.

Canfield, D. E., Jr. and R. W. Bachmann. 1981. Prediction of total phosphorus concentrations, chlorophyll a, and Secchi depths in natural and artificial lakes. Can. J. Fish. Aquatic Sci. 38:414-423.

Carlson, R. E. 1977. A trophic state index for lakes. Limnol. Oceangr. 22:361-369.

Chapin, J. D. and P. D. Uttormark. 1973. Atmospheric contributions of nitrogen and phosphorus. Tech. Rep. Wat. Resources Ctr. Univ. Wis. 73-2, 35pp.

Chapra, S. C. 1975. Comment on "An empirical method of estimating the retention of phosphorus in lakes" by W. B. Kirchner and P. J. Dillon. Water Resourc. Res. 11:1033-1034.

Cornett, R. J. and F. H. Rigler. 1980. The areal hypolimnetic oxygen deficit: An empirical test of the model. Limnol. Oceanogr. 25:672-679.

Dillon, P. J. 1974. A critical review of Vollenweider's nutrient budget model and other related models. Water Res. Bull. 10:969-989.

Dillon, P. J. and W. B. Kirchner. 1975. Reply. Water Resourc. Res. 2:1035-1036.

Dillon, P. J. and F. H. Rigler. 1974a. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19:767-773.

Dillon, P. J. and F. H. Rigler. 1974b. A test of a simple nutrient budget model predicting the phosphorus concentration in lakes. J. Fish. Res. Bd. Can. 31:1771-1778.

Edmondson, W. T. 1961. Changes in Lake Washington following an increase in the nutrient income. Verh. Int. Ver. Limnol. 14:167-175.

Gilbert, R. O. 1987. Statistical Methods for Environmental Pollution Monitoring. Van Nostrand Reinhold Co. Inc., New York NY, 320 pp.

Graneli, W. and D. Solander. 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. Hydrobiologia. 170:245-266.

Henderson-Sellers, B. and H. Markland. 1987. Decaying Lakes: The Origins and Control of Cultural Eutrophication. John Wiley & Sons, Inc., New York NY, 254 pp.

Hodges, J. L. and E. L. Lehman. 1956. The efficiency of some nonparametric competitors of the _t_-test. Ann. Math. Statist. 27:324-335.

Horne, A. J. 1979. Nitrogen fixation in Clear Lake, California. 4. Diel studies on _Aphanizomenon_ and _Anabaena_ blooms. Limnol. Oceanogr. 24:329-341.

Horne, A. J. and G. E. Fogg. 1970. Nitrogen fixation in some English lakes. Proc. Roy. Soc. London. 175:351-366.

Horne, A. J. and C. R. Goldman. 1972. Nitrogen fixation in Clear Lake, California I. Seasonal variation and the role of hereocysts. Limnol. Oceanogr. 17:678-692.

Hutchinson, G. E. 1938. On the relation between oxygen deficit and the productivity and typology of lakes. Int. Rev. Gesamten Hydrobiol. Hydrogr. 36:336-335.

Hutchinson, G. E. 1952. The biogeochemistry of phosphorus. _In_ L. F. Wolterink. ed. The Biology of Phosphorus. Michigan State College Press, Ann Arbor MI, pp. 1-35.

Hutchinson, G. E. 1957. A Treatise on Limnology. I. Geography, Physics, and Chemistry. John Wiley & Sons, Inc., New York NY, 1015 pp.

Hutchinson, G. E. 1973. Eutrophication. The scientific background of a contemporary practical problem. Amer. Sci. 61:269-279.

Jones J. R. and R. W. Bachmann. 1976. Prediction of phosphorus and chlorophyll levels in lakes. J. Water Poll. Control Fed. 48:2176-2182.

Jones J. R. and R. W. Bachmann. 1978. Phosphorus removal by sedimentation in some Iowa reservoirs. Verh. Int. Ver. Theoret. Angnew. Limnol. 20:1576-1580.

Keeney, D. R., R. L. Chen, and D. A. Graetz. 1971. Importance of denitrification and nitrate reduction in sediments to the nitrogen budgets of lakes. Nature 233:66-67.

Kennedy, R. H., K. W. Thornton, and R. C. Gunkel. 1982. The establishment of water quality gradients in reservoirs. Can. Water Resour. J. 7:71-87.

Ketelle, M. J. and P. D. Uttormark. 1971. Problem Lakes in the United States. U. S. Environmental Protection Agency, Univ. Wisconsin, Madison WI.

Kimmel, B. L. and A. W. Groeger. 1984. Organic matter supply and processing in lakes and reservoirs. NALMS Proceedings in Lake and Reservoir Management U. S. EPA 440/5/84-001.

Kirchner, W. B. and P. J. Dillon. 1975. An empirical method of estimating the retention of phosphorus in lakes. Water Res. 11:182-183.

Larsen, D. P. and H. T. Mercier. 1976. Phosphorus retention capacity of lakes. J. Fish. Res. Board Can. 33:1742-1750.

Lehman, J. T. 1980. Release and cycling of nutrients between planktonic algae and herbivores. Limnol. Oceanogr. 25:620-632.

Likens, G. E. ed. 1972. Nutrients and Eutrophication. The Limiting Nutrient Controversy. Special Symposium. Amer. Soc. Limnol. Oceanogr. 1,325 pp.

Lind, O. T. 1985. Handbook of Common Methods in Limnology. Second ed. Kendall/Hunt Pub. Co., Dubuque IA, 199 pp.

Lindeman, R. L. 1942. The trophic-dynamic aspect of ecology. Ecology 23:399-418.

Megard, R. O., J. C. Settles, H. A. Boyer, and W. S. Combs, Jr. 1980. Light, Secchi disks, and trophic states.

Limnol. Oceanogr. 25:373-377.

Mood, A. M. 1954. On the asymptotic efficiency of certain non-parametric two-sample tests. Amer. Math. Statist. 25:514-522.

Mortimer, C. H. 1971. Chemical exchanges between sediments and water in the Great Lakes-speculations on probable regulatory mechanisms. Limnol. Oceangr. 16:387-404.

National Academy of Sciences. 1969. Eutrophication: Causes, Consequences, Correctives. Nat. Acad. Sci., Washington D. C., 661 pp.

Naumann, E. 1919. Nagra synpunkter angaende limnoplaktons okologi med sarskild hansyn till fytoplankton. Svensk Bot. Tidskr., 13:129-163.

O'Neill, J. G. and J. F. Wilkinson. 1977. Oxidation of the effects of ammonia on methane oxidation. J. Gen. Microbiol. 100:407-412.

Ohle, W. 1956. Bioactivity, production, and energy utilization of lakes. Limnol. Oceanogr. 1:139-149.

Oklahoma State Health Department. 1982. Survey of Grand Lake. Special Publication.

Oklahoma Water Resources Board. 1990. Oklahoma Water Atlas. OWRB. 360 pp.

Peters, R. and D. Lean. 1973. The characterization of soluble phosphorus released by limnetic zooplankton. Limnol. Oceanogr. 18:270-279.

Phillips, R. D., H. P. Hotto, and J. C. Loftis. 1988. WQStat II Users Manual. Colorado State Univ., Fort Collins CO, 42 pp.

Porcella, D. B., S. A. Peterson, and D. P. Larsen. 1980. Index to evaluate lake restoration. Procs. ASCE., J. Env. Eng. Div. pp. 1151-1169.

Rast, W. and G. F. Lee. 1978. Summary analysis of the North American (U.S. portion) OECD-Eutrophication Project: Nutrient-loading response relationships and trophic state indices. U.S. Environ. Protect. Agency Rep. EPA 600/3-78-008. 455 pp.

Rawson, D. S. 1939. Some physical and chemical factors in the metabolism of lakes. <u>In</u> Problems of Lake Biology. Publ. Amer. Assoc. Adv. Sci. 10:9-26.

Rawson, D. S. 1955. Morphometry as a dominant factor in the productivity of large lakes. Verh. Int. Ver. Limnol. 12:164-175.

Reckhow, K. H. 1979. Uncertainty applied to Vollenweider's phosphorus criterion. J. Water Pollut. Cont. Fed. 51:2123-2128

Reckhow, K. H. 1988. Empirical models for trophic state in southeastern U.S. lakes and reservoirs. Water Res. Bull. 24:723-734.

Reckhow, K. H. and S. C. Chapra. 1983. Engineering Approaches to Lake Management, Volume I Data Analysis and Empirical Modeling. Buttersworth, Boston MA, 340 pp.

Sakamoto, M. 1966. Primary production of phytoplankton community in some Japanese lakes and its dependence on lake depth. Arch. Hydrobiol. 62:1-28.

Sawyer, C. N. 1947. Fertilization of lakes by agricultural and urban drainage. J. New England Water Works Assoc. 61:109-127.

Schindler, D. W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. Science. 184:897-899.

Schindler, D. W. 1977. Evolution of phosphorus limitation in lakes. Science. 195:260-262.

Schindler, D. W. 1978. Predictive eutrophication models. Limnol. Oceanogr. 23:1080-1081.

Shannon, E. E. and P. C. Brezonik. 1972. Eutrophication analysis: a multivariate approach. J. San. Eng. Div. 98:37-57.

Smith, C. S. and M. S. Adams. 1986. Phosphorus transfer from sediments by Myriophyllum spicatum. Limnol. Oceanogr. 30:1312-1321.

Steel, R. and J. Torrie. 1980. Principles and Procedures of Statistics: A Biometrical Approach. McGraw Hill, New York NY, 633 pp.

Strickland, J. D. and T. R. Parsons. 1972. A practical handbook of seawater analysis. Second ed., Bull. Fish. Res. Board Can. 167, 310 pp.

Strom, K. M. 1931. Fetovatn: A physiographic and biological study of a mountain lake. Arch. Hydrobiol. 22:491-536.

Stumm, W. and J. Leckie. 1971. Phosphate exchange with sediments: Its role in the productivity of surface waters. In S. H. Jenkins [ed.] Advances in water pollution research. Pergamon Press, Oxford. pp. 1-16.

Talling, J. F. 1960. Self-shading effects in natural populations of a planktonic diatom. Wetter u. Leben 12:235-242.

Thienemann, A. 1926. Der Nahrungskeislauf im Wasser. Verh. Dtsch. Zool. Ges. 2:29-79.

Thienemann, A. 1928. Der Sauerstoff im eutrophen und oligotrophen. Die Binnengewasser, 4, 175 pp.

Thornton, K. W., R. H. Kennedy, J. H. Carroll, W. W. Walker R. C. Gunkel, and S. Ashby. 1981. Reservoir sedimentation and water quality-A heuristic model. In H. G. Stefen, ed. Proceedings of the symposium on surface water impoundments. Amer. Soc. Civil Engr., New York NY. pp. 654-661.

Thornton, K. W., B. L. Kimmel, and F. E. Payne. 1990. Reservoir Limnology: Ecological Perspectives. John Wiley & Sons, Inc., New York NY, 246 pp.

Toetz, D. W. 1990. Loading model predictions of total phosphorus in an offset reservoir. Lake Reservoir Manage. 6:181-186.

Toetz, D. and B. Cole. 1980. Ammonia mineralization and cycling in Shagawa Lake, Minnesota. Arch. Hdrobiol. 88:9-23.

United States Environmental Protection Agency. 1977. Survey Report on Grand Lake O' the Cherokees, Ottawa, Mayes, Delaware, and Craig Counties, Oklahoma. Working Paper No. 589. Environ. Monitoring and Support Lab., Las Vegas NV, and Corvallis Environ. Res. Lab., Corvallis OR, 81 pp.

Vallentyne, J. R. 1974. The algal bowl-lakes and man. Ottawa, Misc. Special Publ. 22, Dep. of the Environment, 185 pp.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

Vollenweider, R. A. 1968. Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication. Organisation for Economic Cooperation and Development. DAS/CSI/68.27, 192 pp.

Vollenweider, R. A. 1969. Possibilities and limits of elementary models concerning the budget of substances in lakes. Arch. Hydrobiol. 66:1-36.

Vollenweider, R. A. 1975. Input-output models, with special reference to the phosphorus loading concept in limnology. Schweiz. Z. Hydrol. 37:53-84.

Vollenweider, R. A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33:53-83.

Walker, W. W. 1985. Empirical methods for Predicting Eutrophication in Impoundments; Report 3, Phase II: Model Refinements. U. S. Army Engineer Waterways Exp. Sta., Vicksburg MS, Technical Report E-81-9.

Weber, C. A. 1907. Aufbau und vegetation der Moore Norddentschlands. Beilb. Bot. Jahrb., 90:19-34.

Wetzel, R. G. and G. E. Likens. 1979. Limnological Analyses. W. B Saunders Co., Philadelphia PA, 357 pp.

Wetzel, R. G. 1983. Limnology. Second ed. Saunders College Publishing, Philadelphia PA, 767 pp.

Zar, J. H. 1984. Biostatistical Analysis. Second ed. Prentice-Hall, Inc., Englewood Cliffs NJ, 718 pp.

APPENDIX

CUSTOMIZED BASICA PROGRAM USED FOR ESTIMATING RESTORATION POTENTIAL OF NUTRIENT REDUCTION STRATEGIES

```
10  REM PROGRAM WRITTEN TO SIMULATE RECKHOW'S 1988 RESERVOIR
    MODEL
20  REM WRITTEN BY NOBLE JOBE, III
30  REM 12 AUGUST 1991
40  REM WATER QUALITY RESEARCH LAB - OKLAHOMA STATE
    UNIVERSITY
50  REM USER INPUT OF VARIABLES
60  GOSUB 680
70  INPUT "HYDRAULIC RESIDENCE TIME (YR) = ";TAUW
80  INPUT "MEAN DEPTH (m) = ";ZAVG
90  INPUT "NUMBER OF CHLOROPHYLL SAMPLES = ";NCA
100 PRINT:PRINT
110 PRINT "IF EXTREME OUTLIERS AFFECT THE NORMALITY OF THE
    DATA SET (N OR P), THEN"
120 PRINT "USE QUARTILE DISTRIBUTIONS OF THE DATA.  IF
    OUTLIERS ARE NOT AFFECTING THE"
130 PRINT "NORMALITY OF THE DATA SET USE MINIMUM AND MAXIMUM
    CONCENTRATIONS."
140 PRINT:PRINT:INPUT "DO YOU WISH TO USE (1) MIN/MAX OR (2)
    25th/75th AS THE INPUT P POPULATION";PPOP
150 GOSUB 680:ON PPOP GOTO 190, 170
160 GOTO 140
170 INPUT "1st QUARTILE OF P-INPUT (mg P/l) = ";P25
180 INPUT "3rd QUARTILE OF P-INPUT (mg P/l) = ";P75:GOTO 210
190 INPUT "MINIMUM P CONCENTRATION (mg P/l) = ";PMIN
200 INPUT "MAXIMUM P CONCENTRATION (mg P/l) = ";PMAX
210 GOSUB 680:INPUT "DO YOU WISH TO USE (1) MIN/MAX OR (2)
    25th/75th AS THE INPUT N POPULATION";NPOP
220 GOSUB 680:ON NPOP GOTO 260, 240
230 GOTO 210
240 INPUT "1st QUARTILE OF N-INPUT (mg N/l) = ";N25
250 INPUT "3rd QUARTILE OF N-INPUT (mg N/l) = ";N75:GOSUB
    680:GOTO 280
260 INPUT "MINIMUM N CONCENTRATION (mg N/l) = ";NMIN
270 INPUT "MAXIMUM N CONCENTRATION (mg N/l) = ";NMAX:GOSUB
    680
280 INPUT "PERCENT REDUCTION OF PHOSPHORUS = (ENTER 0 FOR NO
    REDUCTION, 10 FOR 10%, ETC.)";PREDUCE%
290 INPUT "PERCENT REDUCTION OF NITROGEN = (ENTER 0 FOR NO
    REDUCTION, 10 FOR 10%, ETC.)";NREDUCE%
310 TPTOT=0: NTOT=0: CHLORAMAXTOT=0: SDTOT=0: PROBBGTOT=0:
    PROBANTOT=0
320 FOR I = 1 TO 365
330 REM DIRECTS CALCULATIONS FOR CONSTANTS
340 ON PPOP GOSUB 690, 730
350 ON NPOP GOSUB 770, 810
360 PIN=PIN*(1-PREDUCE%/100)
370 NIN=NIN*(1-NREDUCE%/100)
380 KP=3*PIN^(.53)*TAUW^(-.75)*ZAVG^(.58)
```

```
390 KN=.67*TAUW^(.75)
400 REM CALCULATES DISCRIMINANT FUNCTION OF BLUE-GREEN
    DOMINANCE
410 Y=LOG(10)
420 DFBG=4.43-2.05*(LOG(NIN))/Y+3.09*(LOG(PIN))/Y+1.14*
    (LOG(TAUW))/Y
430 REM CALCULATES DISCRIMINANT FUNCTION OF ANOXIC
    HYPOLIMNION
440 DFAN=1.9+.459*(LOG(NIN))/Y+2.21*(LOG(PIN))/Y+1.14*
    (LOG(TAUW))/Y+1.67*(LOG(ZAVG))/Y
450 REM CALCULATES TROPHIC STATE VARIABLES
460 TP=INT((PIN/(1+KP*TAUW))*1000+.5)/1000
470 N=INT((NIN/(1+KN*TAUW))*1000+.5)/1000
480 CHLORAMAX=INT(10^(1.314+.321*(LOG(TP))/Y+.384*
    (LOG(N))/Y+.45*(LOG(NCA))/Y+.136*(LOG(TAUW))/Y)*
    1000+.5)/1000
490 SD=10^(-.47-.364*(LOG(TP))/Y+.102*(LOG(TAUW))/Y+.137*
    (LOG(ZAVG))/Y):SD=INT(SD*1000+.5)/1000
500 PROBBG=1/(1+EXP(-DFBG)):PROBBG=INT(PROBBG*1000+.5)/1000
510 PROBAN=1/(1+EXP(-DFAN)):PROBBG=INT(PROBBG*1000+.5)/1000
520 REM FORMATS OUTPUT
540 IF I = 1 THEN PRINT "     TP  ";"      N  ";"CHLORAMAX
    ";"   SD  ";"  PROBBG ";" PROBAN";" SIGN-BG ";"SIGN-AN"
550 PRINT USING "  #.###  ; #.### ;  ###.#  ; ###.## ; #.##
    ; #.## ; +#.# ; +#.#"; TP; N; CHLORAMAX; SD; PROBBG;
    PROBAN; SGN(DFBG); SGN(DFAN)
560 TPTOT=TPTOT+TP:NTOT=NTOT+N:CHLORAMAXTOT=CHLORAMAXTOT+
    CHLORAMAX: SDTOT=SDTOT+SD: PROBBGTOT=PROBBGTOT+PROBBG:
    PROBANTOT=PROBANTOT+PROBAN
570 NEXT I
580 PRINT "AVERAGE TP                                      =
    ";TPTOT/365
590 PRINT "AVERAGE N                                       =
    ";NTOT/365
600 PRINT "AVERAGE CHLORAMAX                               =
    ";CHLORAMAXTOT/365
610 PRINT "AVERAGE SECCHI DISK DEPTH                       =
    ";SDTOT/365
620 PRINT "AVERAGE PROBABILITY OF BLUE-GREEN DOMINANCE =
    ";PROBBGTOT/365
630 PRINT "AVERAGE PROBABILITY OF HYPOLIMNETIC ANOXIA  =
    ";PROBANTOT/365
640 FOR I = 1 TO 10:PRINT:NEXT I
650 INPUT "RUN AGAIN (Y/N)";REPEAT$
660 IF REPEAT$="Y" THEN 10 ELSE SYSTEM
670 END
680 CLS:FOR I = 1 TO 10:PRINT:NEXT I:RETURN
690 REM CALCULATES INFLUENT P CONCENTRATION USING MIN/MAX AS
    DISTRIBUTION
700 PIN=RND*(PMAX-PMIN)+PMIN
710 PIN=INT(PIN*1000+.5)/1000
720 RETURN
730 REM CALCULATES INFLUENT P CONCENTRATION USING 25th/75th
    AS DISTRIBUTION
```

```
740 PIN=RND*(P75-P25)+P25
750 PIN=INT(PIN*1000+.5)/1000
760 RETURN
770 REM CALCULATES INFLUENT N CONCENTRATION USING MIN/MAX AS
    DISTRIBUTION
780 NIN=RND*(NMAX-NMIN)+NMIN
790 NIN=INT(NIN*1000+.5)/1000
800 RETURN
810 REM CALCULATES INFLUENT N CONCENTRATION USING 25th/75th
    AS DISTRIBUTION
820 NIN=RND*(N75-N25)+N25
830 NIN=INT(NIN*1000+.5)/1000
840 RETURN
```

VITA

Noble Jobe, III

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Master of Science

Thesis: EVALUATION OF THE TROPHIC STATUS OF GRAND LAKE O' THE CHEROKEES WITH REFERENCE TO NUTRIENT MANAGEMENT STRATEGIES

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