RISK BASED TROPHIC STATUS ANALYSIS
IN RESERVOIRS

By

NOBLE JOBE, III

Bachelor of Science
East Central University
Ada, Oklahoma
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Oklahoma State University
Stillwater, Oklahoma
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Dean of the Graduate College

PREFACE

During a flight to Kansas City on November 8, 1993, I took a brief rest from reading and glanced downward. Located directly beneath the flight path, I saw the confluence of the Neosho and Spring rivers with Grand Lake and was inspired by its immense complexity. Then I realized the irony. After all the scientific articles and personal work on this and similar basins, it was this different perspective combined with the two articles in front of me, Garrett Hardin's *The Tradegy of the Commons* (1968) and G. M. Woodwell's *Success, Succession, and Adam Smith* (1974), that made me aware of the enormity of the problems in maintaining environmental quality in reservoir/hydropower systems. Were it not for these papers, perhaps I still would not appreciate the depth of problems in preserving these systems.

I now realize there may not be a single technical solution that will solve this paradigm of problems completely but the scope of solution will include a change in lifestyle of many people, whether "mutually coerced" or not. At 35000 ft in the air, the potential ruin brought about by the commons and constant struggle for ever-decreasing available energies were clear in spite of the immense complexity induced by pollution, recreation, and controlling energy resources. I now question whether the system can be perpetuated and managed accordingly or if the "tradegy of the commons" is a far greater force and premature ruin is inescapable.

In either case, it is not my suggestion to abandon technical aspects of natural resource protection in large mainstem reservoirs because a looming disaster will be the likely result of any shared resource. I believe people's awareness of the ruin is a function of the degree of destruction; increasing ruin induces increasing awareness and diligence for correctives. Unfortunately, history indicates a lag time in which humans remain blind until damage is almost, if not, irreparably severe. Here, I believe, is where science and technology have filled and will continue to fill a critical niché and be called upon to restore and conserve the remaining recoverable and renewable resources. Imperative to this future mission will be a more thorough understanding of how and why the resource was compromised or ruined, not who is responsible. This is where a better understanding of ecosystem dynamics of the natural resource is requisite. It is my intention to provide such insights in this study by illustrating the intrinsic and extrinsic risks of anthropogenic influences on a large mainstem reservoir, Lake Tenkiller, Oklahoma.

Concomitant to an increased appreciation of the problems inherent in conservation, I have been fortunate to gain an exceptional respect for the importance of natural resources and owe much of its credit to my advisor Dr. Bud

Burks. He allowed me time for personal endeavors while encouraging and maintaining high academic standards. The opportunity to balance these two has served as a pivotal element in my success and personal satisfaction during my tenure at OSU. I extend thanks for his guidance, patience, and mentorship. His confidence and support is greatly appreciated. The financial assistance provided also is appreciated. I offer gratitude to Dr. Jerry Wilhm for serving on my committee and providing encouragement, a comfortable environment in which to pursue my studies, and valuable teaching in the classroom. His critical reviews of this project and personal conversations were crucial to my program.

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I also acknowledge Talya Henderson and Jean Bolden of Oklahoma State University Department of Zoology for their assistance, encouragement, and friendship during my degree program.

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I extend a personal note of thanks to my daughters, Richay and Gloria, and my wife, Diane, for their patience

and understanding of my absence while completing my degree. They are a source of indescribable inspiration.

Finally, I thank my father, Noble Jobe, Jr. and mother, JoAnn Bacon for their uncompromising encouragement and unconditional support not only during my education, but my entire life. Without their support, my education would not have been possible.

While I must grant the credits to those mentioned above, I take sole responsibility for any errors or omissions.

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INTRODUCTION

Rationale

In the early to mid 20^{th} century, concern for flood control, soil erosion, and water resources in the southern United States, an area devoid of natural lakes, prompted construction of many reservoirs along established river channels. These reservoirs were designed to serve many purposes, but three major uses prevail: 1) control soil erosion which provides better opportunities for agricultural development; 2) flood control; and 3) provide water storage for municipalities and hydroelectric power generation, recreation, and public water supplies. Scientific *status quo* during the construction "boom" ostensibly did not afford adequate foresight of long term water quality trends in these impoundments. More focus was placed on the reservoir's potential effects on the river. Ironically, due to large economic developments and unforeseen benefits around large reservoirs, an increased focus on current and long-term reservoir water quality problems now prevails while construction of new reservoirs has ebbed.

Whether or not reservoir construction was appropriate is moot; these reservoirs now provide essential economic bases for local communities and the general public. Therefore, preserving the integrity of these systems not only further advances environmental protection of the ecosystem, but also sustains the lifestyles of many people who rely on them.

Although water quality problems in reservoirs range widely, the two most common seem to be sedimentation and cultural eutrophication. As this study will illustrate, the two are really intimately coupled and thus actually represent a single problem in large mainstem reservoirs.

As in natural lakes, reservoir eutrophication has been linked to increased nutrient loads. These loads are derived from municipal sewage outfalls and agricultural runoff which impinges the waterway from diffuse sources, commonly called nonpoint source (NPS) pollution, which originate in the watershed. In fact, many studies have shown a large portion of NPS nutrient loads in mainstem reservoirs originate from agricultural practices (Novotny and Chesters 1981). There is profound irony here; increased agricultural production in fertile floodplains afforded by flood and erosion control was one of the justifications for reservoir construction.

While arguing benefits and costs of preserving the integrity of reservoirs for agricultural practices seems intuitive, several researchers believe a harmonic compromise exists that protects the value of the reservoir and enhances agricultural production. Although neither may be maximized individually, to deduce and effectuate a balance requires extensive holistic approaches that include evaluation of land use effects on basin export of nutrients and pollutants, lotic transport, reservoir response, and integration of economic impact considerations to these paradigms. Such holistic approaches require teams of specialists each investigating a given integral component. This study illustrates such a component in an Environmental Protection Agency (EPA) Clean Lakes Program (CLP) Phase I Diagnostic/Feasibility study on Lake Tenkiller, Oklahoma by assessing the response of the reservoir to allochthonous phosphorus loads and to investigate the structural and functional dynamics of the intrinsic and extrinsic risks of eutrophication-related symptoms in the mainstem reservoir.

Scope of Work

Reservoir managers must weigh benefits of economic growth and stability and feasibility of management scenarios against potential environmental impacts during the decision-making process while accommodating public demand. Within this framework, it is imperative the manager/agency communicate these options to the public in a coherent manner.

To reduce environmental and ecological complexity and uncertainty, resource managers and agencies are now shifting to assessing risks, or probability of an adverse effect, posed by anthropogenic influences, the goal of which is to maximize benefits while minimizing risks. However, the traditional approaches in risk assessments have been developed on toxicological studies directed towards human health, e.g., hazardous waste sites, while recent risk assessments now appear to have pervaded environmental protection for the conservation of ecological systems, often called ecological risk assessment (ERA).

I believe a limnologist's role in this framework is to assess risks of potential environmental effects (costs) on the stream and/or reservoir, under a certain anthropogenic influence, whether a subsidy, as in nutrients, or stress, as in a toxic contamination. However, preliminary to this assessment is a requisite knowledge of the ecosystem's function and how it is related to the system's sensitivity to an extrinsic influence and the desired goal. If scientists cannot elucidate these properties disastrous policy may result (see Calow 1994). For example, an assessment of reservoir eutrophication requires the knowledge that increased nutrients promote undesirable conditions, e.g., algal blooms and an increased productivity which induces a larger oxygen demand which promotes hypolimnetic anoxia during summer stratification. This conceptual foundation of the eutrophication process and its responses have been researched extensively, illustrated many times, and eventually led to the ban of phosphate-based detergents in the Great Lakes region in April 1972. The agreement among limnologists on the above framework of the eutrophication process is indicated by general acceptance of classic indicators, such as trophic state indices, critical nutrient levels, etc., all of which have been deterministic.

Application of comparative ecological risk assessment techniques in eutrophication assessment and management has not been realized. Also, concurrent to the development of ERAs came a focus on larger scale projects, i.e., watershed projects and basin management plans. I believe these two directions occurring simultaneously is fortuitous and provides a direction for development of mainstem reservoir eutrophication risk models.

Lake Tenkiller, located in eastern Oklahoma, provides an example of this approach and serves as the focus of this study. Recreational uses of Lake Tenkiller include angling, boating, camping, swimming, and various aquatic sports. Self-contained-underwater-breathing-apparatus (SCUBA) and snorkel divers also are attracted to Lake Tenkiller due to its exceptional clarity. The Illinois River's compromised clarity (EPA 1991), and the lake's excessive nutrient loads (Nolen et al. 1989), exemplify the typical "reservoir problem" and thus provides a unique study site with the necessary conditions for initial development and evaluation of a comparative risk analysis model that describes mainstem reservoir eutrophication.

With intensively growing agricultural practices (especially poultry production) in a drainage basin that is characterized by rocky substrate with cracks and fissures, Lake Tenkiller appears prone to continued cultural eutrophication; it represents the "best lake in the worst place" situation. In this case, any benefits of lake protection could and probably will be at the cost of land users in the basin. Therefore, Lake Tenkiller further provides an opportunity to develop compromises such as those discussed previously where risk analysis bridges risk management and communication. Essential to success of such a compromise, however, is a thorough understanding of reservoir ecosystem structure and function and its resilience to allochthonous influences, i.e., nitrogen and phosphorus loads in eutrophication. I developed a risk-based model that describes the trophic status dynamics of reservoirs by integrating the link between the abiotic (causes) and biotic (responses) components while accommodating trends in spatial and temporal heterogeneities (fate and transport) in the reservoir. More specifications are discussed later.

Novel approaches to eutrophication assessment and management in reservoirs are needed because historical research on natural lakes, while providing some insights, rely on assumptions that are violated when applied to reservoirs. Most of these violations center on spatial and temporal heterogeneities in reservoirs, which appear to follow a predictable pattern that is quite different than in natural lakes (Thornton et al. 1990). These heterogeneities in reservoirs are thought to be dictated largely by inlake hydraulic phenomena, e.g., flow, which are determined by lake morphometry and allochthonous hydraulic loads/releases, i.e., runoff and volume management of the reservoir. This pattern is the basis for describing the intrinsic risks of the reservoir to a nutrient loading regime and thus serves as the foundation of the eutrophication risk model presented in this study.

I believe that reservoirs have two risk elements, extrinsic and intrinsic. The extrinsic risk element is hydraulic regimes (i.e., inflow/outflow), due to its stochasticity, its linkage with phosphorus and nitrogen loads, and its effect on the longitudinal gradient of water quality in the reservoir. I am not asserting "flow causes eutrophication"; merely the impact of nutrient loads which co-occur with hydraulic events will be determined by the spatial gradients of nutrient processes within the reservoir which are dictated by hydraulics.

I believe the intrinsic risk element of a reservoir's eutrophication to a given allochthonous nutrient load can be assessed by the longitudinal gradient of nutrient fate and transport dynamics. Thus, a model that simulates the reservoir's response to allochthonous nutrient loads as determined by the inlake hydraulic regimes could be useful in stochastic simulations with the largest element of uncertainty being hydraulic events. Using techniques (and results) from standard flood risk analyses coupled with abiotic loads and biotic responses in a deterministic model of reservoir eutrophication afford stochastic assessment or the probability distribution of the reservoir's response domain. This association is the foundation of the reservoir ecological risk factor (RERF) model.

The objectives of this study were to:

1) identify and characterize statistical distributions of spatial trends in phosphorus, nitrogen, turbidity, and transparency in Lake Tenkiller,

- calculate temporal and longitudinal kinetics in decay of the above parameters in Lake Tenkiller
- estimate the temporal and spatial chlorophyll *a* gradients in Lake Tenkiller

2) calculate a hydraulic budget of Lake Tenkiller

3) develop a risk-model that describes the intrinsic risk of eutrophication of Lake Tenkiller by evaluating the reservoir ecological risk factors (RERF)

4) integrate the reservoir ecological risk factors to evaluate trends of trophic status of Lake Tenkiller,

5) evaluate trends in trophic status of Lake Tenkiller under various nutrient load reduction scenarios and predict concomitant eutrophication responses

6) recommend a management strategy and propose a monitoring schedule.

The null hypotheses were:

H_o: Lake Tenkiller does not exhibit predictable temporal or spatial trends in eutrophication-related parameters (i.e., nitrogen, phosphorus, turbidity, transparency, and chlorophyll density)

H_o: hydraulic regimes are not correlated with the spatial or temporal trends in Lake Tenkiller

H_o: no ecological trophic status improvement (i.e., oligotrophication) is predicted from simulating nutrient reduction plans.

CHAPTER II

LITERATURE REVIEW

Causatives of Eutrophication

Since the conception of labelling lakes according to trophic status by Naumann (1919), research on the dynamics of eutrophication has been exhaustive. The conclusion of most of this research indicated increased phosphorus loads accelerate eutrophication, especially when the phosphorus loads are disproportionate to nitrogen loads. In North America, two classic examples are Lake Washington directed mostly by Walles Edmondson and the Experimental Lakes Area (ELA) project directed mostly by David Schindler. These studies are detailed below because the Lake Washington study illustrated long-term trends of accelerated eutrophication and its reversal, while the ELA project applied a more scientifically-controlled method of demonstrating the effects of increased phosphorus loads. Coincidentally, these classic studies were both conducted ca. late 1960 to early 1970, when Richard Vollenwieder (1968) concurrently developed quantitative relationships between phosphorus loads and eutrophication. Vollenwieder's model and subsequent modifications are discussed later.

More in-depth reviews on theoretical principles of eutrophication can be found in Beeton and Edmondson (1972), Hutchinson (1973), Schindler and Fee (1974), Welch (1976), and Lee et al. (1978).

Lake Washington

Lake Washington, located near Puget Sound in Washington, received two sewage contaminations, one in the early 1900's for which sparse data exists and one from the early 1940's to early 1960's. Walles Edmondson's work on Lake Washington started in 1955 and focused on the latter episode of sewage contamination (Edmondson 1961, Edmondson 1966, Edmondson 1970, Edmondson 1972, Edmondson et al. 1956, Edmondson and Lehman 1981).

Lake Washington received increasing inputs of domestic sewage from 1941 to 1963 (Edmondson and Lehman 1981). From 1964 to ca. 1968, treated sewage was diverted into Puget Sound and thus bypassed Lake Washington. The decrease was about 75,700 m^3 d^{-1} to near zero by 1968 (Edmondson 1972, Edmondson and Lehman 1981).

Detailed nutrient budgets prior to diversion were based on data from 1957 and 1964 and post-diversion budgets were calculated from an extensive monitoring program (Edmondson and Lehman 1981). These budgets showed concomitant decreases in phosphorus loads (mostly total phosphorus) of approximately 200,000 kg P yr^{-1} in the mid 1960's to approximately 50 kg P yr^{-1} post-diversion, or > 99% (Edmondson and Lehman 1981). Pre-diversion blue-green dominance was as high as 90% and decreased by 1977-1978 to a maximum dominance less than 20%. However, the most dramatic effect was seen in chlorophyll densities which prior to diversion exhibited values as high as ≈ 60 μg/l with a high variance (presumably from blooms and population crashes) to a more stable seasonal oscillation at ≈ 5-10 μg/l (Edmondson and Lehman 1981). The apparent variances of mass loadings and chlorophyll *a* densities decreased while the variance of lake phosphorus content did not (Figures 6 and 7 in Edmondson and Lehman 1981).

Experimental Lakes Area

The effects of phosphorus on eutrophication was demonstrated by the extensive Experimental Lakes Area (ELA) project and was studied by David Schindler et al. at the Freshwater Institute, Winnipeg, Manitoba. The ELA project originated in 1968 by the Government of Canada and the Province of Ontario (Johnson and Vallentyne 1971) and denotes one of the earliest applications of the scientific method to "whole-lake" experiments on eutrophication of lakes (Schindler 1980). The ELA is located in northwestern Ontario and includes several lakes previously undisturbed by human influence. These lakes are part of the Canadian Precambrian Shield Lakes with similar characteristics and served as the experimental units in the study. Although the scope of the project included many "whole-lake" questions (e.g., acidification, heavy metal pollution, effects of clearcutting and forest fires, nitrilotriacetate), the primary thrust for which the project is noted are the effects of carbon, nitrogen, phosphorus, and various combinations of the three nutrients on eutrophication. It is this thrust that is relevant to this study and hence is the only aspect explained here.

Traditionally, the causative factors of eutrophication have been studied using the Printz Bottle Test nutrient limitation assays in which monocultures, usually *Selenastrum capricornutum*, are spiked with the above nutrients (and various combinations) and the respective growth rates monitored. The only difference in the nutrient addition experiments in the ELA study and the bottle test is whole ecosystems are used instead of bottled monocultures. The advantages include elimination of errors due to extrapolation from bottled monocultures to ecosystem effects and use of other limnological parameters as real endpoints instead of inferred endpoints. Accordingly, the respective treatments used in the eutrophication experiments in the ELA project closely resembled those of bottle tests.

Lake 227 (L227) was the first to be used for artificial eutrophication trials and received weekly inputs of 0.34 g P/m^2 and 5.04 g N/m^2 for 17 weeks annually from 1969 to 1974 (Schindler and Fee 1974). Each treatment equated to an elevation of 68 μg N/l and 4.5 μg P/l (natural loading of N and P to Lake 227 were 19-20% and 17-18%, respectively). After the first year of treatment (i.e., 1969), annual chlorophyll maxima increased from 3 μg/l in 1968 to 51.8 μg/l, while chlorophyll densities in the control lakes, 240, 239, 304, 305 ranged from 0.6 μg/l to 8.6 μg/l (Schindler et al. 1971).

Schindler et al. (1971) concluded from the first trial on L227 that the added N and P were quickly assimilated thus causing the increased biomass and decreased CO_2 concentrations. However, measurements of primary productivity did not show significant increases; possible explanations are given by Fee (1979). After 4 years of treatment which increased natural N and P inputs approximately 5-10x and in spite of extremely low carbon content, midsummer standing crop of phytoplankton had increased by two orders of magnitude (L227 $\approx$ 200 μg/l vs. controls $\approx$ 2-5 μg/l) and dominant algal genera shifted from Cryptophytes and Chrysophytes to Chlorophytes and Cyanophytes which included *Oscillatoria*, *Lyngbya*, *Pseudoanabaena*, *Oocystis*, and *Dictyosphearium* (Schindler et al. 1973; Schindler and Fee 1974). The authors postulated the observed biomass denoted the maximum possible under the given light and circulation regimes, i.e., self-shading (for more discussion on self-shading see Talling 1960). Although L227 became anoxic during the ice-free season, no P recirculation was observed and approximately 80% of the incoming P was sedimented via inorganic and biotic sedimentation processes within the year of treatment (Schindler 1973). Researchers concluded that carbon was not a causal factor in eutrophication because L227 was abnormally low in carbon and approximately 69-95% of the necessary carbon for the observed algal biomass was from the atmosphere. Hence, carbon did not hinder eutrophication if N and P were supplied (Schindler et al. 1973).

When nutrient fertilization regimes were stopped, the lake recovered quickly, ostensibly from rapid turnover of P fractions. This recovery provided insights for limnologists who maintained that rate of P supply is a better predictor of eutrophication potential than absolute P concentrations (see Vollenweider 1968 and Vollenweider 1975). Schindler and Fee (1974) proposed that for incoming P to accelerate eutrophication, it must be available in the euphotic zone which was smaller than the epilimnion in the ELA. Thus, eutrophication might be curtailed by discharging P loads into the hypolimnion. Lake 302 (L302) was chosen for this experiment because it had two regions between which flow was restricted, thus providing experimental (north) and control (south) sections. Fertilization of the north basin with 0.54 g P, 2.79 g N, and 3.73 g C m^{-2} yr^{-1} for 21 weeks (May-Oct) by hypolimnetic injection did not produce algal blooms as did epilimnetic fertilization schemes in other lakes. Although a small fall bloom occurred in the north basin, it was not different than in the south basin and was much less than blooms produced from epilimnetic fertilization schemes (Schindler and Fee 1974).

Although results from the L227 trial strongly suggested that atmospheric recruitment of carbon was commensurate with N and P loads, the algal community showed signs of carbon shortage when chlorophyll densities exceeded ca. 100 μg/l (Schindler and Fee 1974). Therefore, to test the effects of anthropogenic carbon inputs on accelerated eutrophication, Lake 304 (L304) was fertilized with 0.40 g P, 5.2 g N, and 5.5 g C m^{-2} yr^{-1} (values expected in sewage outfalls) and phytoplankton standing crops were monitored. Results suggested eutrophication would occur without dissolved inorganic carbon depletion as in L227 but the phytoplankton standing crops were not larger (in fact smaller) than would be expected based on the N and P inputs exclusively (Schindler and Fee 1974).

After eutrophication of L304 was assessed, tertiary treatment was simulated by removing the P input while maintaining the N and C fertilization. Phytoplankton decreased rapidly and no bloom occurred in the following year, again illustrating that P was the primary agent that accelerated eutrophication (Schindler and Fee 1974). Fee (1979) later argues nitrogen may have been interacting.

With the controversy of carbon limitation waning, the question of N limitation needed to be resolved. Contemporary thought held that nitrogen deficiencies (whether via P repletion or N depletion) favored N-fixing algae, e.g., Cyanophytes. With the ELA's critics arguing that interlake comparisons of dominant algal genera could be due to different original algal compositions and not nutrient loading regimes, Schindler et al. (1971) placed polyethylene tubes in L227 and spiked individual series with C, N, P, and various combinations of C, N, and P. The results of these trials suggested, as interlake comparisons did, that increased carbon had no effect and nitrogen deficiencies (via P repletion in the tubes) induced eutrophication and promoted blue-green algal dominance, *Anabaena* in L227 (Schindler et al. 1971, Schindler 1977). These results suggested a nitrogen recruitment capacity similar to carbon but specific to those algae capable of fixing molecular N, again implicating, albeit indirectly, P as the primary cause of accelerated eutrophication.

Another lake (L261) that received only P fertilization showed an increase in phytoplanktonic chlorophyll density but not a dominance of phytoplanktonic N-fixing algae (Schindler 1977). However, the lake increased in N-fixing periphyton and a lower chlorophyll:productivity ratio as would be expected in accelerated eutrophication (Schindler 1977).

A follow-up study on this phenomenon was instigated on Lake 226 (L226) which, like L304, had a narrow restriction between two large sections of the lake. A curtain was installed at the point of constriction to separate two portions of the same lake (north and south basins). Initial conditions were identical. Fertilization schemes were 0.6 g P, 3.2 g N, and 6.1 g C m^{-2} yr^{-1} for the north basin and 3.2 g N and 6.1 g C m^{-2} yr^{-1} for the south basin (Schindler and Fee

1974). The north basin which received the P, N, and C loads developed an immense algal bloom (mostly *Anabaena*) while the south basin which received only N and C did not develop blooms (Schindler and Fee 1974).

The remaining issue to be resolved was that of atomic ratios and the development of algal blooms. Sakamoto (1966) proposed that absolute P concentration is not a good predictor of eutrophication. He proposed that the amount of N relative to P indicates the relative N deficiency and thus would serve as a better predictor of phytoplanktonic dominance of N-fixers, i.e., bluegreens. Specifically, a total N:total P ratio $< \approx$ 10-15 would favor N-fixers. Schindler (1977) tested this hypothesis with L226. Prior to the 1975 fertilization regimes of L227, the lake had a total N:total P ratio of $\approx$ 14 and the dominant genera of algae were *Scenedesmus* (a green alga) and non-N-fixers. When total N:total P in L226 was $\approx$ 5 and *Anabaena* (a bluegreen N-fixer) dominated (Schindler 1977). If Sakamoto (1966) is correct, decreasing the total N:total P ratio in L227 should shift algal dominance to N-fixers. So, in 1975, the total N:total P ratio in L227 was reduced to 5 (to duplicate that of L226). As a result, *Aphanizomenon gracile*, another known N-fixing blue-green alga, became dominant for the first time in 8 years (Schindler 1977). These results clearly illustrated merits of N:P ratios in algal community regulation and again implicated P loads as the primary causal agent of eutrophication.

A summary of the ELA project conclusions include:

- Although lakes undergoing eutrophication may exhibit signs of C shortage, atmospheric recruitment accommodates this deficiency.
- Continuous elevated P loads are required to induce and maintain accelerated eutrophication.
- Controlling P inputs exclusively can alleviate symptoms of eutrophication; the "added P" will eventually be lost to the sediments.
- Controlling N is futile because atmospheric recruitment via N-fixation will accommodate any deficiency. In fact, controlling N may exacerbate eutrophication because it creates conditions favorable for bluegreen algal dominance.

Based upon the findings of the Lake Washington, the ELA studies, and elevated P levels in Lake Erie, the United States and Canada signed a water quality agreement in 1972 effectively banning the addition of phosphates in detergents used in the region. Some people questioned the validity of such legislative mandates because results suggested N-limitation in highly eutrophic systems (Anonymous 1971), thus implying N control as the potential corrective action. This dogma is undoubtedly a result of misunderstanding the Lake Washington and ELA studies which clearly showed N-limitation was a sign of transition induced and maintained by excess P loads. As a result of the debate, science progressed and most limnological entities promoted P control as the appropriate corrective action for accelerated eutrophication. The scientific concurrence of this management approach and the advent of computers stimulated development of quantitative models in assessing criteria for external P loads that lead to eutrophication problems. This modelling approach has continued over 2 decades.

Quantitative Analysis of Eutrophication

Loading Models

Phosphorus

Most of the traditional methods have been based on or modified from Vollenweider's (1968) nutrient loading concept and attempts to predict inlake P concentrations under observed allochthonous nutrient loads. Since the models proposed in this project do not rely on inherent assumptions common to the traditional loading models, only a brief discussion of these models is presented (for a more detailed description of traditional models or their derivations consult the original citations or Jobe 1991).

Originally proposed for lakes by Biffi (1963), Vollenweider's (1968) loading model is based upon a mass balance equation of P inputs and outputs,

$$V \frac{dP}{dt} = M_P - Q P - \sigma P V \qquad \textbf{(1)}$$

where P is the inlake P concentration, t is time, V is the lake volume, M_P is the impinging annual P load, Q is the annual hydraulic load (outflow volume), and σ is the net sedimentation coefficient. To predict inlake P concentration, the derivative must be set to zero (this is the steady state assumption or dP/dt=0) and solve for P,

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

where P is inlake P concentration, z_a is the mean depth, L is the areal loading (i.e., M_P/Area), σ is the net sedimentation coefficient, and ρ is the hydraulic flushing rate (Q/V and often denoted as τ_w^{-1}). Vollenweider (1968) used this approach to delineate trophic status boundaries of lakes on a plot of L versus z_a/τ_w, while recognizing the only parameter that can be managed was L. Therefore, acceptable limits of P loads in a given lake would be determined by its z_a and hydraulic load. The reason for including z_a/τ_w was that lakes with larger volumes and smaller residence times could tolerate larger P loads without eutrophication symptoms.

The major criticism of Vollenweider's model was the estimation of σ. Therefore, with the model given in equation 2 as the theoretical construct, some investigators empirically estimated σ, while others attempted to estimate inlake P from measured external loads by choosing to estimate a P retention coefficient (Dillon and Rigler 1974a, Larsen and Mercier 1976, Canfield and Bachmann 1981, Reckhow 1988), which describes the fraction of inflowing P retained by the lake (Table I). Some of these models differentiated natural and artificial lakes, while others were developed exclusively for natural lakes. Ironically, as suggested by the results of the Lake Washington and ELA studies, a lake's response has been found to be dependent on the magnitude of the P flux, and not the concentration. Since, criteria for trophic status have been based upon measured inlake P concentration, the goal of these models was to estimate the resulting inlake P concentration.

These models, having the mass balance equation as the theoretical basis, assume the lake is a continuously stirred tank reactor (CSTR) and is in steady state (i.e., $dP/dt = 0$). Although long time frames in natural lakes may approximate this assumption and short-lived reservoirs do not, these models have been applied to artifical lakes with some success (Larsen and Mercier 1976, Canfield and Bachmann 1981, Reckhow 1988). Large run-of-the-river reservoirs violate the CSTR assumption by differential P sedimentation rates in the headwaters when compared to the lower reaches (Thornton et al. 1990). This differential sedimentation rates are influenced by seasonal hydraulic regimes (Thornton et al. 1990). The reservoir ecological risk factor (RERF) model conserves the steady state assumption but does not assume a CSTR. It is this advantage that provides a better comparative indicator among and within reservoirs while incorporating an assessment of intrinsic risk.

Nitrogen

During the peak of eutrophication research ca. late 1960 to ca. late 1970s, controversy surrounded the acceptance of accepting phosphorus as the primary cause of accelerated eutrophication. Most of this controversy

Table I. Traditional Phosphorus Models in Lakes.

Reference	Model Equation	CSTR Assumption
Vollenweider (1968)	inlake $P = L/(z_a(\sigma+\rho))$	Yes
Dillon and Rigler (1974a)	inlake $P = L(1-R_{exp})/(z_a\rho)$	Yes
Vollenweider (1975)	$\sigma = 10/z_a$	Yes
Kirchner and Dillon (1975)	$R_P = 0.426e^{-.271q} + 0.574\ e^{-0.00949q}$	Yes
Larsen and Mercier (1976)	$R_{exp} = 1/(1+\rho_w)$	Yes
Canfield and Bachmann (1981)	$\sigma_{natural\ lakes} = 0.162(L/z_a)^{0.458}$	Yes
	$\sigma_{artificial\ lakes} = 0.114(L/z_a)^{0.589}$	Yes
	inlake $P = fL/(z_a(\sigma+\rho))$	Yes
Reckhow (1988)	$\log(P) = \log[P_i/(1+k_P\tau_w)]$	Yes
	$k_P = 3.0P_i^{0.53}\tau_w^{-0.75}z_a^{0.58}$	
	$\log(N) = \log[N_i/(1+k_N\tau_w)]$	Yes
	$k_N = 0.67\tau_w$	
	$\log(ch)_{max} = 1.314+\log(P^{0.321}N^{0.384}n_c^{0.45}\tau_w^{0.136})$	
	$\log(Secchi) = -0.47+\log(P^{-0.364}\tau_w^{0.102}z_a^{0.137})$	

stemmed from data that suggested algal growth as being controlled by carbon or nitrogen in advanced stages of eutrophication (Bowen 1970, Anonymous 1971). The resulting debate ended with dispelling the carbon school of thought while the nitrogen versus phosphorus continues today. Thus, many nitrogen indices of eutrophication have been proposed and merit discussion.

Most nitrogen-based trophic status indices have been based upon inlake nitrogen concentrations instead of a loading model. However, exceptions exist. Baker et al. (1985) proposed a simple modification to Vollenweider's (1968) P loading model by incorporating nitrogen loading as the primary nutrient in the mass balance equation for Florida lakes. Using this model, Baker et al. (1985) predicted more accurate chlorophyll *a* levels in the Florida lakes as well as 101 National Eutrophication Survey (NES) lakes. They proposed the decision of which model to use (i.e., the N or P) should depend upon which gives the lowest chlorophyll *a* prediction (Baker et al. 1985). I assume the rationale for such a decision was that lower values reflected more extreme limitation and thus more closely followed the lake's trend in eutrophication.

Trophic State Indices

To simplify comparison among lakes, assess current trophic status, and monitor effectiveness of restoration schemes, several empirical "inlake" indices have been developed. The salient strengths and weaknesses of each are discussed below.

Carlson's (1977) trophic state index (TSI) traditionally has been the most popular and is based upon the regression of natural log transformed values of spring surface total phosphorus (μg P/l), Secchi disk depth (m), and chlorophyll *a* (μg/l). Three TSIs can be calculated from these three data, 1) TSI(CHL), 2) TSI(TP), and 3) TSI(SD). He scaled Secchi disk limits of 0 and 64 m to TSI values of 100 and 0, respectively, where increasing TSI values denoted increasing eutrophy (Carlson 1977). A TSI increase of 10 units equated to a halving of the Secchi disk and doubling of surface P. Chlorophyll *a* was the dependent variable and thus the model did not impose such linearity. Computational forms of each TSI are:

$$TSI\ (CHL) = 10\left(6 - \frac{2.04 - 0.68\ \ln\ (chl)}{\ln 2}\right) \tag{3}$$

$$TSI\ (TP) = 10\ \left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right) \tag{4}$$

$$TSI\ (SD) = 10\left(6 - \frac{\ln SD}{\ln 2}\right) \tag{5}$$

Carlson's (1977) models have been used widely and apply mostly to natural lakes. These TSIs assume P-limited algal growth and algal biomass controls transparency. Reservoirs may temporarily exhibit the first but varies from transparency controlling biomass (headwaters, i.e., light limited growth) to algal biomass controlling transparency in the lacustrine zone (Thornton et al. 1990). This trend presents a problem in correlation studies and denotes uncertainty in directionality. Thus, whole-lake correlation studies in reservoirs provide little insight into the system process. Significant deviations among Carlson's (1977) three TSI's from the headwaters to the lacustrine zone in reservoirs most likely result from this phenomena. Therefore, a biased assessment of reservoir trophic status results from using such whole-lake indices of homogeneity. However, Jobe (1991) illustrated that these deviations could be minimized if the headwater stations were eliminated and only zones where the algae were not light-limited were included. The weakness of this approach is the resulting assessment only describes the lacustrine zone.

Carlson (1980b) and Osgood (1982b) have proposed using TSI deviations on natural lakes to describe lake typology (e.g., argillotrophic, dystrophic), but warns the TSI is an index not a definition of trophic status, i.e., the index and its deviations reflect trophic structural conditions not a functional process like accelerated eutrophication. Carlson (pers. com.) further believes the trophic status of a lake, by definition, is a manifestation of lake function or response of a lake not its structure, *per se*. The RERF integrates all zones of the reservoir as a continuum and compensates for the spatial transition from light to nutrient limitation.

Canfield (1983) proposed a similar model that predicts chlorophyll *a* concentrations from observed inlake nitrogen concentrations. This index is more applicable to southern lakes that are highly eutrophic. Canfield (1983) proposed the nitrogen-based predictor was more appropriate for eutrophic systems because algal growths in these lakes were characteristically N-limited. Predicted chlorophyll *a* in this model was given by:

$$\log (Chl\ a) = -0.15 + 0.744 \log (TP), \quad r^2 = 0.59 \tag{6}$$

$$\log (Chl\ a) = -2.99 + 1.38 \log (TN), \quad r^2 = 0.77 \tag{7}$$

$$\log (Chl\ a) = -2.49 + 0.269 \log (TP) + 1.06\ (TN), \quad r^2 = 0.81 \tag{8}$$

The best correlation from the above parameters was obtained from inclusion of TN and TP.

Kratzer and Brezonik (1981) also developed a trophic state index based on nitrogen for several Florida lakes. Their computational form was:

$$TSI\ (TN) = 54.45 + 14.43 \ln (TN) \tag{9}$$

where TN was total nitrogen (mg N/l). Kratzer and Brezonik (1981) suggested, similar to Canfield's (1983) approach, to consider the TSI(TN) with Carlson's (1977) TSI(TP) and average the lower of the two with Carlson's TSI(SD) and TSI(CHL) to derive a TSI(AVG). This averaging protocol was criticized by Osgood (1982a) and Lambou (1982), while Kratzer and Brezonik (1982), Brezonik and Kratzer (1982), and Baker et al. (1985) defended the TSI(TN) usage for the same reasons. Although both sides agreed the study lakes on which the TSI(TN) was developed were N limited, arguing for or against incorporation of the N parameter in trophic state indices parallels the nitrogen versus phosphorus debate.

Shannon and Brezonik (1972b) used a multivariate analysis (cluster analysis) to develop two multiparameter TSI's that included a variety of eutrophication-related parameters. This index categorized colored and clear lakes. The rationale for a multivariate approach was given by Brezonik (1984). Computational forms, respectively, were:

$$TSI_{Colored} = 0.848 \frac{1}{SD} + 0.809\, COND + 0.887\, TON + 0.768\, TP + 0.930\, PP + 0.780\, CHA + 0.893 \frac{1}{CR} + 9.33 \quad (10)$$

$$TSI_{Clear} = 0.936 \frac{1}{SD} + 0.827\, COND + 0.907\, TON + 0.748\, TP + 0.938\, PP + 0.982\, CHA + 0.579 \frac{1}{CR} + 4.76 \quad (11)$$

where SD is Secchi disk transparency (m), COND is conductivity ($\mu\mho$/cm), TON is total organic nitrogen (mg N/l), TP is total phosphorus (mg P/l), PP is primary productivity (mg C/m^3-hr), CHA is chlorophyll *a* (mg/m^3), and CR is the cation ratio (Na + K)/(Ca + Mg) (Shannon and Brezonik 1972b). The TSI boundaries that denoted transitions from oligotrophy to mesotrophy to eutrophy were $\approx$ 1.2 and $\approx$ 5, respectively. Although the Shannon and Brezonik (1972b) TSIs are useful from an empirical perspective, ecological oversimplification coupled with the cost of obtaining the necessary data has hindered its widespread application. Also, this TSI, like most single indices, reduces the analysis of trophic status to a single "whole lake" criterion based solely on structural properties; it describes nothing of a lake's functional responses.

Porcella et al. (1980) proposed a similar multi-parameter index that incorporated scalar transformations of Carlson's (1977) TSI's among other trophic state parameters and coined it the lake evaluation index (LEI). The LEI was developed to evaluate restoration effectiveness and was calculated as the arithmetic average of the individual scalars where the TP parameter or TN scalar were mutually exclusive according to which yielded the lowest LEI. As argued by Baker et al. (1985), Kratzer and Brezonik (1982), and Brezonik and Krazter (1982), the reason for choosing the lower was that the lower parameter was in lesser supply and thus most likely denoted the limiting nutrient, thereby reflecting the eutrophication trend. The computational form was:

$$LEI = 0.25\,[0.5\,(XCA + XMAC) + XDO + XSD + XTP] \tag{12}$$

where

$$XCA = 30.6 + 9.81 \ln(CA), \qquad XCA \leq 100 \tag{13}$$

$$XMAC = PMAC, \qquad XMAC \leq 100 \tag{14}$$

$$XDO = 10 \frac{\left(\sum_{i=0}^{i=Z_{MAX}} |EDO - CDO| \Delta V_i \right)}{V} \tag{15}$$

$$XSD = 60 - 14.427 \ln (SD), \qquad XSD \leq 100 \tag{16}$$

$$XTP = 4.15 + 14.427 \ln (TP), \qquad XTP \leq 100 \tag{17}$$

$$XTN = 14.427 \ln (TN) - 23.8, \qquad XTN \leq 100 \tag{18}$$

and EDO and CDO represented equilibrium dissolved oxygen concentrations predicted from temperature profiles and current (or measured) DO concentrations for stratum z, respectively. The XDO scalar was omitted from the LEI equation in the original published equation, but the accompanying text describes its incorporation. Due to the original textual description and since the multiplier was 0.25 (implying an arithmetic average of n = 4), I assume the exclusion of XDO was merely an oversight and thus have included it in the above LEI formulation. Analogous to Carlson's (1977) scale, the LEI ranges from 0 (denoting extreme oligotrophy) to 100 (denoting extreme eutrophy). Weaknesses in the LEI, some of which Porcella et al. (1980) recognized, include:

1) individual scalars for a given lake may exhibit large discrepancies,

2) arithmetic averaging will result in the same LEI if the discrepancies of the individual scalars offset one another,

3) applies a single index of homogeneity and thus

denotes a "whole" lake structural descriptor, and

4) gives equal weights to a multitude of eutrophication symptoms when fewer actually may exist or exceed one another in significance (e.g., deep eutrophic lakes are not likely to have a macrophyte problem thus inclusion of XMAC is knowingly biased).

The LEI and the other trophic state indices discussed above can be used, however, to evaluate functional aspects of lakes. Some authors proposed using changes or deviations in the indices over time to assess trophic status rather than absolute values (Carlson 1980b, 1983). Unfortunately, reservoirs present both functional responses in time and a strong spatial heterogeneity that imposes unique considerations in using such singular "whole-lake" indices of homogeneity or their deviations. The intent of using the RERF is to accommodate such heterogeneity.

Finally, two functional models that have been used as trophic state indicators are areal and volumetric hypolimnetic oxygen depletion rates, AHOD and VHOD. However, these models assume hypolimentic oxygen depletion is caused by community respiration (mostly bacterial) of endogenic organic material, material which was solely produced in the epilimnion or at least a constant fraction of it (Hutchinson 1938). Significant variance was observed when these models were applied to natural lakes, but most problems seem to have stemmed from differing fractional hypolimnetic volumes. Reservoirs may approximate natural lakes in the constancy of fractional hypolimnetic volumes but have hypolimnetic oxygen dynamics that are highly dependent on inflowing oxygen demand and hydraulic regimes (i.e., short hypolimnetic τ_w). These influences can decrease the AHOD, while allochthonous organic loads can increase the AHOD. Clearly, these phenomena influence the impact of epilimnetic productivity (or a constant fraction of it) on AHOD and VHOD rates (Thornton et al. 1990). For these reasons, AHOD rates have been of little use in assessing trophic status of reservoirs. I am not suggesting hypolimnetic oxygen levels lack diagnostic significance. On the contrary, hypolimnetic anoxia not only indicates increasing eutrophy, it exacerbates the problem by P recycling. I merely suggest numerical criteria for reservoir AHOD and VHOD are of little use. For further reviews on AHOD and VHOD model details, computation, and interpretation, the reader is referred to Hutchinson (1938), Cornett and Rigler (1979), Welch (1979), and Cornett and Rigler (1980).

Reservoirs

Reservoirs have introduced an array of unique questions on accelerated eutrophication to limnologists. Traditional approaches have applied classical limnological concepts to reservoirs which are usually qualitatively valid. However, when quantitative methods are used, large errors often result. These problems have led contemporary thought to differentiate reservoirs and natural lakes as distinct ecosystems.

The differences between the two types of ecosystems ostensibly are derived from the external influences upon the ecosystem's behavior and the system's internal response patterns. Typical glaciated lakes are influenced by diffuse allochthonous nutrient loads but have a high degree of autochthonous control mechanisms with a large degree of homogeneity. In contrast, reservoirs are driven by less diffuse allochthonous loads (i.e., point of entry is the headwaters) and maintain allochthonous control in the headwaters but exhibit a gradation to autochthonous control near the lower end (dam) indicating more heterogeneity. Reservoirs exhibit characteristics similar to rivers in the headwaters and similar to glaciated lakes near the dam, but not enough of each to be classified as either type of system (Table II). Assessment of eutrophication-related processes in large mainstem reservoirs must accommodate these phenomena. Accurate modelling of these processes must account for the uniqueness of these "hybrid" ecosystems.

Longitudinal Gradients

The major abiotic influences on reservoirs originate in the watershed from point source and nonpoint (diffuse) sources. The stream network serves to focus the loads and impinges upon the reservoir at the headwaters. These loads vary most during the spring and fall when runoff occurs more frequently and with greater variation. These events tend to be processed in a "plug-flow" manner instead of a diffusive

Table II. Comparison of a River, Reservoir, and Glacial Lake (modified from Thornton et al. 1990).

Attribute	River	Reservoir	Glacial Lake
Watershed influence	greater	intermediate	lesser
Shoreline	elongate	astatic	circular
Water level	variable	variable	natural
Flushing rate	rapid	intermediate	slow
Ionic composition	variable	intermediate	stable
Sedimentation	low	high	low
Turbidity	high	intermediate	low
Organic accumulation	low	rapid	slow
Nutrient supply	allochthonous	both	autochthonous
Nutrient loss	advection	both	sedimentation
Growth selection	rapid (r)	intermediate (r & K)	homeostatic (K)
Immigration/ extinction	rapid	rapid	slow
Spatial structure	longitudinal gradients	both	vertical gradients

homogenization throughout the ecosystem as in natural lakes.

During summer, the runoff, lake level, and incoming nutrient loads stabilize and form a longitudinal gradient in which a water quality trend is established along a continuum. Along this continuum, the headwaters are dominated by qualities of the incoming hydraulic load (river) which shifts to lacustrine conditions near the dam (Table III).

The gradient is determined by hydraulic flow characteristics in the reservoir. In the headwaters, flow velocity is sufficient to keep inflowing sediments in suspension in the water and few differences between river water and lake water exist. The velocity of the influent water decreases as the width and depth of the reservoir increases causing a clarification as suspended solids sediment. As depth of the euphotic zone increases and available nutrients are still relatively high, a peak in algal growth occurs. This is the zone of transition (Thornton et al. 1990). The velocity of water continues to decrease which allows finer particles to settle and gives way to a quiescent, less variable body of water. This zone maintains characteristics most similar to natural lakes and hence has been coined the lacustrine zone (Thornton et al. 1990). A comparative summary is given in Table III.

Modelling The Trend

Quantification of the spatial trend consists of describing the longitudinal gradient as a continuum of change along the thalweg of the reservoir. Intrinsic parameters that drive the system appear to be the availability of light (i.e., turbidity or transparency) which increases and phosphorus which decays with thalweg

Table III. Characteristics of the Longitudinal Gradient in Large Mainstem Reservoirs (modified from Thornton et al. 1990).

Factor	Riverine	Transition	Lacustrine
Basin	narrow	intermediate	broad, deep
Flow Velocity	high	reduced	low
Transparency	low	intermediate	high
Turbidity	high	intermediate	low
Light Availability	$z_{photic}<z_{mix}$	increased	$z_{photic}>z_{mix}$
Nutrients	high	intermediate	low
Primary Productivity	light limited	high	nutrient limited
Organic Matter Supply	external	intermediate	internal
Trophic Status Trend	eu-	eu/meso-	oligo-

distance. The endpoint used in this model is chlorophyll *a* density, although other biotic variables, e.g., algal biomass, diversity, algal cell counts, could be used and should be investigated.

Driving Variables. While few models have been fit to the trend in light availability, total phosphorus has been shown to exponentially decay with distance (Thornton et al. 1990). Motulsky (1987) gives the generalized exponential,

$$TP = A\,e^{(-TD)} + B \qquad \textbf{(19)}$$

where, TP is the total phosphorus concentration at thalweg distance TD, A is the decay coefficient, and B is an asymptote at which TP decay is 0. While the model's simplicity is attractive, it forces the inital decay of TP to occur at the upper bound of the reservoir (i.e., thalweg distance = 0). Data from Lake Tenkiller illustrate that TP decay may start as far as 30% downstream along the thalweg. Therefore, because reservoirs are known to change the TP/turbidity decay according to seasonal hydraulic regimes (i.e., spring or fall runoff), a more appropriate model for accommodating such trends is a sigmoidal equation which can evaluate various decay rates at any location of the transition zone.

This precept is supported further by the length/cross-sectional area relationship of a pyramid. If a large mainstem reservoir approximates a pyramid laying on its side the cross-sectional area increases with distance (height) as a point moves from its apex to base. It follows that because velocity is a quotient of flow divided by cross-sectional area and flow is constant (or nearly so) in a mainstem reservoir, the increase in cross-sectional area should equate to a decrease in velocity. At some point inherent to the morphometry of the reservoir (i.e., dimensions of the pyramid), velocity slows to a point at which the suspensoids begin to settle; further decreases in velocity would have little effect. Hence, a sigmoidal trend in environmental correlates of suspended particles, e.g., P coprecipitates, can be expected. Applying Motulsky's (1987) sigmoidal equation to TP on distance yields,

$$TP_i = MIN + \frac{MAX - MIN}{1 + B\ e^{SF(EL_{50} - TD)}} \tag{20}$$

where, TP_i is total phosphorus concentration (μg P/l) at thalweg distance *i*, MAX is maximum TP observed for the given data, MIN is minimum TP observed, SF is the slope factor which describes the rate at which TP decays, EL50 is the effective length (thalweg distance) at which TP decay is 50% complete, and TD is the thalweg distance.

A different approach is suggested for turbidity. Unlike phosphorus, a decreasing turbidity increases algal growth until light is replete (the transition zone). Therefore, an inverse relationship of turbidity and algal growth is expected. Applying Motulsky's (1987) sigmoidal equation to turbidity on distance yields,

$$TB_i = MIN + \frac{MAX - MIN}{1 + B\ e^{SF(EL_{50} - TD)}} \tag{21}$$

For light-related measurements that indicate transparency (e.g., Secchi disk), the same model can be applied. If Secchi disk is applied, a transformation of the depth to decimeters is suggested because the typical Secchi disk depths in reservoirs usually are 10 m or less. I have rarely observed Secchi disk depths > 10 m in Grand Lake. Scaling the Secchi disk depths to decimeters establishes a range that approximates that of turbidity in nephelometric turbidity units (NTUs) and TP in μg P/l. This scaling affords direct comparison of model parameters and coefficients. Applying Motulsky's (1987) sigmoidal equation to Secchi disk depth on distance yields,

$$SD_i = MIN + \frac{MAX - MIN}{1 + B\, e^{SF(EL_{50} - TD)}} \tag{22}$$

where SD is the Secchi disk depth (dm) and the other factors are as previously described relative to the spatial trend in Secchi disk. It is worth noting that much controversy has been generated on the relationship between algal biomass and Secchi disk depths (Tyler 1968, Bannister 1975, Lorenzen 1980, Megard et al. 1980, Carlson 1980a, Edmondson 1980). However, this controversy has been directed towards the effect of chlorophyll and/or algal biomass upon the transparency. In the above model, the opposite is being simulated where the transparency controls the algal biomass in the headwaters via light limitation and undergoes a transition to nutrient limitation upon decay. The point of contention for past controversy is insignificant in this model.

Clearly, the sigmoidal model assumes the highest TP and turdidity will occur at the headwaters and the minimum occurs downstream from the maximum, while opposite transparency trends exist. If this trend is not observed in a given snapshot of data (i.e., a given day), the reservoir doesn't approximate equilibrium and thus invalidates the model. In such a case, the reservoir is probably processing a "plug" of "flow" as described above and hence is in a state of transition.

<u>Response Variables.</u> The longitudinal gradient in chlorophyll density occurs as a maximum at the transition zone. I postulate that the zone of maximal chlorophyll density occurs at a point where light availability increases to a point of saturation for the algae and another environmental factor becomes restrictive (i.e., nutrient limitation). It is this linkage upon which the RERF model is based.

Alternatively, a strong argument for hydraulic control could be made because flow regimes could regulate mixing depth and thus suspension of algal cells, thereby optimizing enviromental conditions for algal cells by preventing settling. However, my hypothesis is based on the close correlation of the occurrence of the chlorophyll maximum with the turbidity and TP decays. It is not likely that the algal community would be adapted to velocity requirements that are optimized so near the range of that for suspensoid and coprecipitate settling, especially given the variety of buoyancy adaptations of algae.

If the assumed linkage is true, a relationship between the driving variables and the endpoint will exist. I propose that the maximum chlorophyll density can be predicted by an equation in which light and TP are optimized. Such a relationship can be evaluated by a variety of equations but I chose a maxima function as given by Spain (1982) of chlorophyll *a* density on thalweg distance. I acknowledge the weakness of this approach is a lack of parametric coefficients that have a theoretical basis. However, the estimated coefficients are evaluated on a relative basis and are not characterized as representing a physical process. The maxima function is represented by,

$$CA = CRF \times TD \times e^{(PF \times TD)} \tag{23}$$

where CA is chlorophyll *a* density (μg/l), CRF is an empirical chlorophyll response factor, TD is thalweg distance (decimal fraction), and PF is an empirical power factor. If the hypothesis is true, a close correlation between kinetics of transparency and phosphorus decay with the chlorophyll maximum should exist.

CHAPTER III

MATERIALS AND METHODS

Study Site

Lake Tenkiller is located in eastern Oklahoma in Cherokee and Sequoyah counties. It is part of the Arkansas River drainage and the dam lies on the Illinois River at river kilometer 20.6 (approximate) about 11.3 km northeast of Gore and 35.4 km southeast of Muskogee, Oklahoma. Construction of the lake, which was authorized under the Flood Control Act of 1938 and River and Harbor Act of 1946, began in 1947 with full flood control operation by 1953. The lake was built by the United States Army Corps of Engineers. The rolled earthfill dam is approximately 923 m long and 61 m above the streambed. Outlet works include ten 15.4 X 7.7 m tainter gates, a 5.8 m diameter hypolimnetic conduit, and a 5.8 m diameter hypolimnetic intake to the powerhouse. Lake morphometric data are given in Table IV.

Water quality sampling sites were chosen according to an estimated gradient indicated by spatial trends in historical data (Nolen et al. 1989). Eight sites were chosen to coincide with the longitudinal gradient (discussed in Chapter II) with more stations located in the zone

Table IV. Lake Tenkiller Morphometry.

Parameter	Value
Elevation (NGVD)	
@Conservation pool	632.0
@Flood pool	667.0
Capacity (km^3)	
@Conservation pool	0.81
@Flood pool	1.52
Area (km^2)	
@Conservation pool	52.2
@Flood pool	84.2
Depth (m)	
Mean	15.5
Maximum	46.3
Relative	0.57
Shoreline length (km)	209
Shoreline development	8.17
Volume development	1.00
Average hydraulic residence time (yr)	0.76

Table V. Geographic Positions of Lake Tenkiller Sampling Sites.

Sampling Site	Latitude (N)	Longitude (W)
1	35°49′14"	94°54′11"
2	35°46′01"	94°53′10"
3	35°45′47"	94°53′31"
4	35°45′23"	94°54′22"
5	35°44′15"	94°57′11"
6	35°40′32"	94°58′35"
7	35°36′13"	95°02′53"
8	35°35′29"	95°03′33"

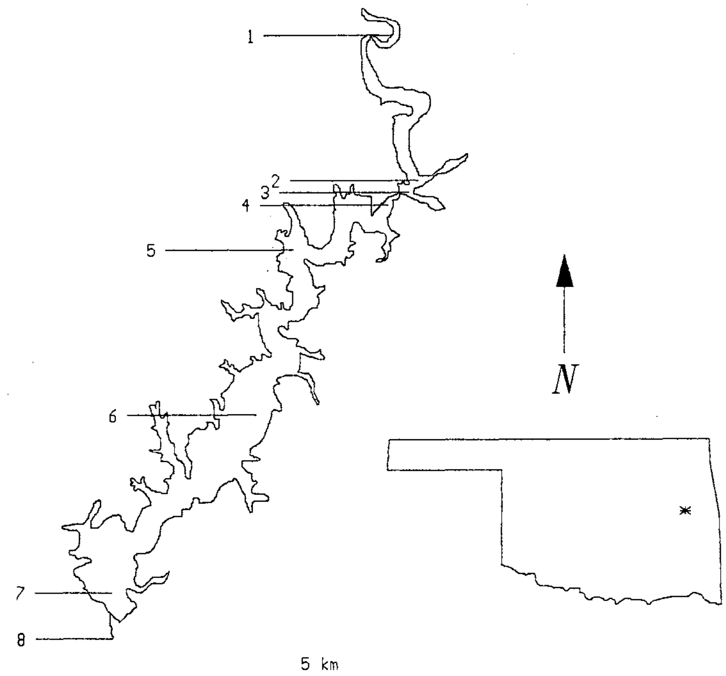

Figure 1. Lake Tenkiller Water Quality Sampling Sites.

of transition and one headwater and tailwater site for hydraulic and nutrient budget assessment (Figure 1).

Water Quality Analyses

Each of the sampling sites were sampled approximately monthly from Apr 92 to Oct 93. During each sampling time, I collected epilimnetic (0.5 m below the surface) and hypolimnetic (0.5 m above sediment) water samples in clear acid-washed high-density polyethylene (HDPE) or polypropylene (PP) bottles and one epilimnetic sample in a non-acid-washed opaque HDPE or PP bottles at each site. The samples were placed on ice and returned to Oklahoma State

University for analyses. The analyses were conducted within 48 hr and included total phosphorus and ortho-P, total and phenolphthalein alkalinity, and total hardness as per Lind (1985), total nitrogen as per Bachmann and Canfield (1990), and chlorophyll *a* as per Standard Methods (APHA 1989). Additional lab analyses included nitrate-N, chloride, and sulfate by liquid chromatography (EPA Method 300.0, Pfaff et al. 1989).

Secchi disk depth, turbidity, pH, and depth profiles of dissolved oxygen, temperature, and conductivity were collected. Depth profiles were measured with Yellow Springs Instrument meters and turbidity was measured by a Hach model 16800 turbidimeter. Secchi disk depth was measured with a standard 20 cm diameter Secchi disk.

Biological data collected included adundance and composition of fish, zooplankton, and algae. Collection of algae was a single grab sample of surface water (0.5 m below surface) preserved in Lugol's solution as per Lind (1985), while zooplankton were collected by complete vertical tows with a standard 80 μm Wisconsin net and preserved in 5% neutral formalin, respectively. Enumeration of algae and zoooplankton were as per Lind (1985). More detail on algal abundance and composition methods may be found in Haraughty (unpub. data) and fish and zooplankton methods in Kloxin (unpub. data). Statistical summaries of water quality parameters were computed and presented as notched box and whisker plots as described by McGill et al. (1978). Although sample size usually is indicated by the width of the box, I chose to use a standard box width and present sample sizes in tabular format (Table VI). These plots illustrate the upper and lower limits, first, second, and third quartiles, and an approximate statistical domain at $\alpha \approx 0.05$ (Figure 2; McGill et al. 1978). Statistical domain is used here to indicate the range at which statistical significance is indicated when two or more domains (i.e., notches) do not overlap.

Hydraulic Budget

A hydraulic budget was estimated based upon 24 hr

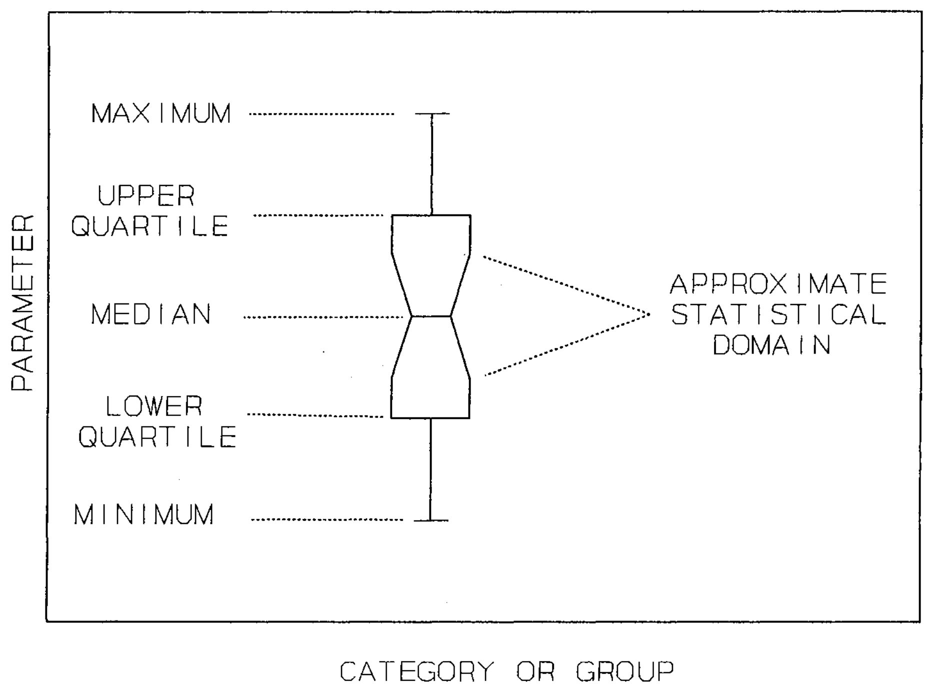

Figure 2. Interpretation of the Notched Box and Whisker Plot as Suggested by McGill et al. (1978).

averages of hourly lake elevation data provided by USACE. General models for elevation/capacity and elevation/area were constructed by nonlinear least squares regression to an exponential model of the form:

$$CAPACITY(ac\text{-}ft) = A*ELEVATION*e^{B*ELEVATION}+E*ELEVATION \tag{24}$$

and

$$AREA(acres) = A*ELEVATION*e^{B*ELEVATION}+E*ELEVATION \tag{25}$$

where, A and B are estimated coefficients and E is an estimated asymptotic coefficient. Capacity data used in the regression were provided by USACE and areal data given in USACE (1993). Hydraulic residence time was computed by dividing the sum of flows at two gauging stations nearest the lake (USGS07196500 - Illinois River near Tahlequah and USGS07197000 - Baron Fork at Eldon) by the estimated capacity for each day. The flows for the stations were downloaded from STORET, while the capacity was estimated based on a 24 hr average of hourly elevation data provided by USACE. Hydraulic balance was assessed by subtracting the inflows (USGS07197000 + USGS07196500) from the outflow (USGS07198000 - Illinois River near Gore).

Risk Model Description

The reservoir ecological risk factor model is based on the longitudinal water quality gradient in mainstem reservoirs as described in Chapter II. I fit existing data for total phosphorus, turbidity, and Secchi disk depths to a sigmoidal function (Equations 20, 21, and 22) using least squares nonlinear regression in SYSTAT©. The spatial trend in chlorophyll density was fit via the same method to a maxima function (Equation 23) also using SYSTAT©. Data were fit to the equations using a decimal-scaled thalweg distance as the independent variable. The purpose of scaling the distance was to standardize for reservoir size, thus enabling comparative analysis with other reservoirs of differing sizes. The model was run on the cumulative data set and separately for each sampling event (i.e., selected by date). The model parameters (defined on p. 37) EL50, MIN, and MAX of Equations 20, 21, and 22 were used to infer intrinsic properties of phosphorus fate, transport, and impact. A correlation matrix was created among the abiotic model parameters (i.e., EL50, MIN, and MAX), estimated chlorophyll maximum and its location, and hydraulic conditions (i.e., inflow, outflow, capacity, and instantaneous residence time (τ_w). From these results, a general model that predicts the spatial gradient from hydraulic conditions was constructed.

The maxima function was fit to the longitudinal trend in chlorophyll *a* density and the resulting estimated parameters, CRF and PF (defined on p. 40), were evaluated. These model parameters with the estimated errors were used as the reference for trophic status. Inference of the trophic status from the model parameters was used because the chlorophyll *a* densities at the various stations indicated the entire range of trophy could be identified. For example, many dates indicated chlorophyll densities > 30 $\mu g/l$ (eutrophy) at the mid stations while the lower stations indicated < 5 $\mu g/l$ (oligotrophy). Using the model parameters allows a correction for the maxima peak (intensity) and area distribution of trophic status.

Since the PF induces a higher peak, CRF influences the peak breadth, and the effect of one is relative to the other, I opted to integrate both parameters in the overall assessment of reservoir trophic status (Figure 3). The dotted lines represent equivalent chlorophyll *a* maxima and have been drawn at 5, 10, 20, 40, and 80 $\mu g/l$. I chose these values because the traditional boundary of oligotrophy and mesotrophy is approximately 5 $\mu g/l$. The remaining lines of Figure 3 represent a doubling of the chlorophyll maxima. The axes limits on Figure 3 were not intended to be absolute. However, if the power factor is 0 or greater the maxima trend cannot be observed within the 1.0 scaled thalweg distance and hence invalidates the model anyway. Chlorophyll densities greater than 80 $\mu g/l$ can produce self-shading and hence induce another inhibitory factor not modelled here (see Talling 1960). If the traditional reference of trophic status is to be inferred, increasing eutrophy is indicated by proceeding right and upward in Figure 3. Boundaries of trophic status depend on subjective judgment.

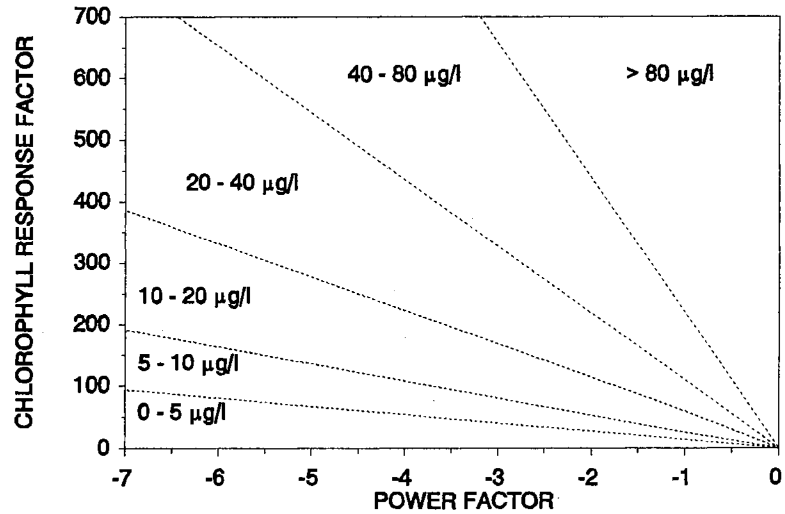

Figure 3. Integration of CRF and PF into Trophic Status Characterization (lines represent isoclines of chlorophyll maxima).

Total Maximum Daily Load

While the model developed here describes the sensitivity of a reservoir to allochthonous loads of phosphorus, external loads must be managed according to the intrinsic sensitivity. Historically, external loads have been calculated on an annual basis. However, recent management strategies have targeted daily loads in lieu of annual loads. I calculated the daily load by dividing the estimated annual load by 365. This total maximum daily load (TMDL) of phosphorus for Lake Tenkiller was calculated by four methods, Vollenweider's (1968, 1975) critical loads, a morphoedaphic index (MEI) method proposed by Vighi and Chiadauni (1985), Reckhow's (1988) model, and applying the assimilative capacity inferred from the RERF where lacustrine conditions were targeted at 0.01 mg P/l for an upper oligotrophic boundary and 0.02 mg P/l the upper mesotrophic boundary (> 0.02 mg P/l indicate eutrophy).

Vollenweider's (1968) model as refined by Vollenweider in (1975, 1976) defined critical levels of areal P loading which he called "dangerous" for the meso/eutrophic boundary and "permissible" for the oligo/mesotrophic boundary in lakes based upon the observed areal hydraulic loading divided by the mean depth. Vollenweider (1976) empirically derived:

$$L\ (P)_{critical} = A * q_s * (1 + \sqrt{\frac{z_{avg}}{q_s}}) \tag{26}$$

where L(P) is the critical areal loading (g P/m^2/yr), A is a coefficient defined by the trophic boundary (10 for oligo/mesotrophic boundary and 20 for meso/eutrophic boundary, q_s is the areal hydraulic loading (m/yr), and z_{avg} is the mean depth. The observed values were calculated and compared to estimated "critical" L(P) values for the observed hydraulic conditions of each sampling event.

A second reference point was calculated based upon the regression of two MEIs (alkalinity and conductivity) on observed inlake P concentration under "no anthropogenic influence" (Vighi and Chiaudani 1985). The equations were:

$$\log[P] = 1.44 + 0.33(\pm 0.10) \log MEI_{alk} \qquad r = 0.87 \tag{27}$$

and

$$\log[P] = 0.71 + 0.26(\pm 0.11) \log MEI_{con} \qquad r = 0.72, \tag{28}$$

where MEI_{alk} and MEI_{con} are total alkalinity(meq/l)/z_{avg} and conductivity (μS)/z_{avg}, respectively. One can argue that no such background P concentration exists for reservoirs because the "anthropogenic construction" of the reservoir implies influence by default. However, no acceptable alternative method of estimating nonanthropogenic background loadings for reservoirs has been evaluated. Therefore, methods given by Vighi and Chiaudani (1985) were calculated and used for comparative purposes only.

Reckhow's (1988) model of trophic state for southeastern reservoirs was used to simulate nutrient reduction effects. In the process, the percent reduction required for a downstream total P concentration of 10 μg P/l was used as an oligotrophic target. From this percent reduction, a distribution of TMDLs was calculated as estimated percent reduction in current daily load. Whether or not the reduction in load equates to a similar reduction in downstream concentration is unknown and should be monitored.

Finally, a total maximum daily load was estimated according to the lakes ability to assimilate the load. In the sigmoidal equation (Equation 20), the difference in the upper and lower asymptotes provides an estimate of the reservoir's assimilative capacity in the transition zone. Based upon this assumption, I calculated a TMDL for total P at the headwaters of Lake Tenkiller that would yield a lacustrine (i.e., station 7) concentration of 0.01 and 0.02 mg P/l for the upper boundaries of oligotrophic and mesotrophic conditions, respectively. These limits are in accordance with currently accepted boundaries as applied to P-limited phytoplankton in natural lakes. I exclusively applied the limits to the lacustrine zone of the reservoir because it most emulates the natural lake ecosystem, most frequently P-limited (Haraughty unpub. data), and represents the portion of the reservoir most sensitive to P-availability. Additionally, Jobe (1991) found an improved agreement among Carlson's (1977) TSIs in Grand Lake, Oklahoma when only data from the lacustrine zone were used (i.e., data from the riverine and transition zones introduced more error among the TSIs than did lacustrine data).

CHAPTER IV

RESULTS AND DISCUSSION

Current Conditions in Water Quality

Statistical summaries of water quality conditions from the 1974 EPA-NES study (USEPA 1977), 1985-86 USACE study (USACE 1988), and the current 1992-93 Clean Lakes Program (CLP) study were compared for temporal and spatial trends. Observed trends indicated a temporal increase in total P from 1974 to 1985 and then a decrease in 1992. Hypolimnetic total P has also increased since 1974, but it is difficult to determine if this is due to an artifact of underflow of higher headwater loads or P recirculation resulting from the apparent accelerated eutrophication. Differences in epilimnetic total N and nitrate N were not statistically significant. Chlorophyll *a* in the midlake region has increased by almost three times since 1974, but apparently did not affect the Secchi disk tranparencies.

Epilimnetic total P indicated the expected sigmoidal trend in the 1985-86 USACE data and the 1992-93 CLP data, but could not be ascertained from the 1974 NES data (Figure 4). Two possiblities exist. The NES study did not include a "critical" headwater station as did the later two studies and did not have as many samples (Table VI). However, an apparent decrease in total P at Horseshoe Bend has occurred since 1985-86 from a median of 209 μg P/l in 1985-86 to 122 μg P/l in 1992-93 which was statistically significant (Figure 4). The most plausible explanation is the city of Tahlequah (located just above the headwaters) implemented a P control sewage treatment plant in the interim.

Also, a comparison of epilimnetic total P among the EPA-NES station 4, USACE station 13, and CLP station 2 (the locations were approximately the same) showed a similar trend. For these stations, the median of 48 μg P/l in 1974 increased to 173 μg P/l in 1985-86 and then decreased to 98 μg P/l in 1992-93. All were statistically significant (Figure 4).

Finally, epilimnetic total P trends at the most downstream station (i.e., EPA-NES station 1, USACE station 1, and CLP station 7) showed a slightly different trend. These data, located near the dam, showed an increase from a median value of 42 μg P/l in 1974 to 98 μg P/l in 1985-86 and then decreased to 24 μg P/l in 1992-93. All were statistically significant (Figure 4).

The sample size of the EPA-NES study was only four. Yet, these four data represented both low-flow and above average flow conditions and thus should have approximated the true range. All median values for all three studies exceeded the suggested surface values of 20 μg P/l during the summer.

Hypolimnetic total P trends showed a similar temporal trend, but also showed more spatial variation, especially at

Table VI. Sample Sizes of Referenced Studies on Lake Tenkiller.

		EPILIMNION						HYPOLIMNION	
Study	Site	TP	TURB	TN	NO_3	SD	CA	TP	TURB
NES	1	4	ND	4	4	4	4	4	ND
	2	4	ND	4	4	4	4	4	ND
	3	4	ND	4	4	4	4	4	ND
	4	4	ND	4	4	4	4	4	ND
USACE	1	10	16	ND	10	13	14	8	9
	2	10	16	ND	10	14	14	7	8
	3	10	16	ND	10	15	14	8	9
	4	10	16	ND	10	13	14	7	8
	5	10	16	ND	10	14	14	7	8
	6	10	16	ND	10	14	14	7	8
	7	10	15	ND	10	14	14	7	7
	8	10	15	ND	1	15	13	5	6
	9	9	15	ND	1	15	13	6	7
	10	9	16	ND	9	15	12	2	2
	11	9	15	ND	9	14	13	3	4
	13	9	15	ND	9	14	13	9	2
	14	9	15	ND	9	14	13	ND	ND
CLP	1	16	14	16	16	1	16	--[1]	--[1]
	2	17	14	17	17	18	22	17	14
	3	17	14	17	18	18	18	17	14
	4	17	14	17	18	18	18	17	14
	5	17	14	17	18	18	22	17	14
	6	17	14	17	18	18	22	17	14
	7	17	14	17	18	19	18	17	14

ND = no data available.
[1] = No stratification observed at Station 1 thus limnetic layers were not differentiated.

the midlake stations (Figure 5). I hypothesize the highest values at the midlake region is a result of P recirculation during stratification. Current dogma contends that anoxia develops first in the transition zone because it has the highest organic load and a lower hypolimnetic volume

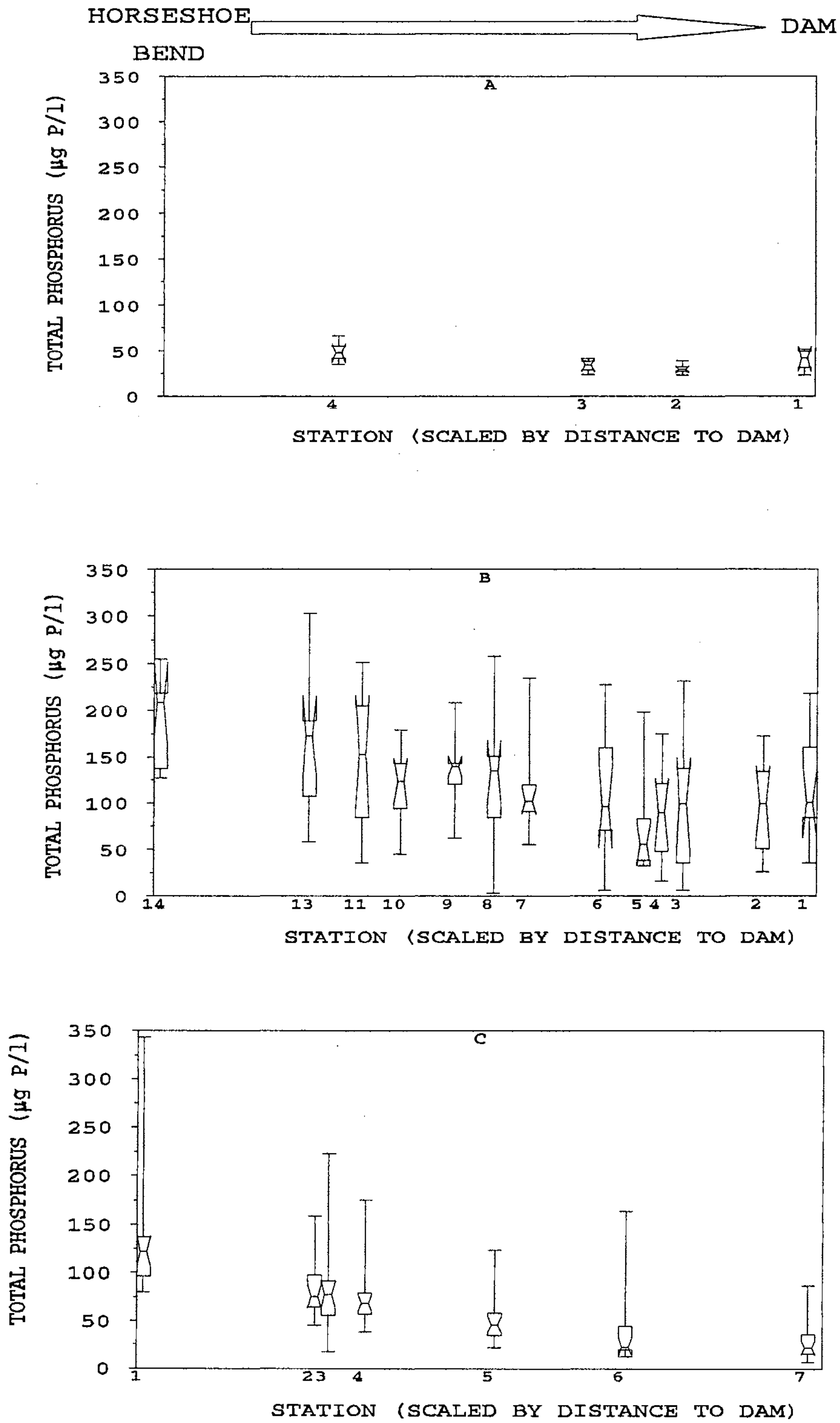

Figure 4. Epilimnetic Total P Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP Study 1992-93.

(Thornton et al. 1990). Therefore, the duration when redox potentials are favorable to P recirculation is longer in this region and thus allows for more accumulation (Thornton et al. 1990). However, none of the midlake stations were significantly different than the headwater station (i.e., Horseshoe Bend).

Although nitrogen fractions were not incorporated into the actual model, the relevance of nitrogen to eutrophication of lakes cannot be overlooked. The common assumption to most, if not all, phosphorus-based eutrophication models and indices is phosphorus-limited algal growth. Sakamoto (1966) found this to be true only for values of TN:TP greater than approximately 12. As discussed in Chapter II, lakes usually show symptoms of nitrogen-limitation as a result of being P replete and will revert to phosphorus-limitation if the phosphorus supply is reduced sufficiently (see Chapter II, Experimental Lakes Area). Critical concentrations have been proposed by Sawyer (1947) as 300 and 600 μg N/l for oligo/mesotrophic and meso/eutrophic boundaries, respectively. Therefore, while nitrogen was not incorporated into the RERF model, the data were evaluated and discussed.

In the CLP study, inorganic nitrogen (nitrate-N) in the epilimnion showed an initial decrease in concentration from a median of 1170 μg N/l at station 1 to 461 μg N/l at station 2 with little decrease through the remaining stations (Figure 6). The USACE study in 1985-86 showed a different trend where the median nitrate N at Horseshoe Bend

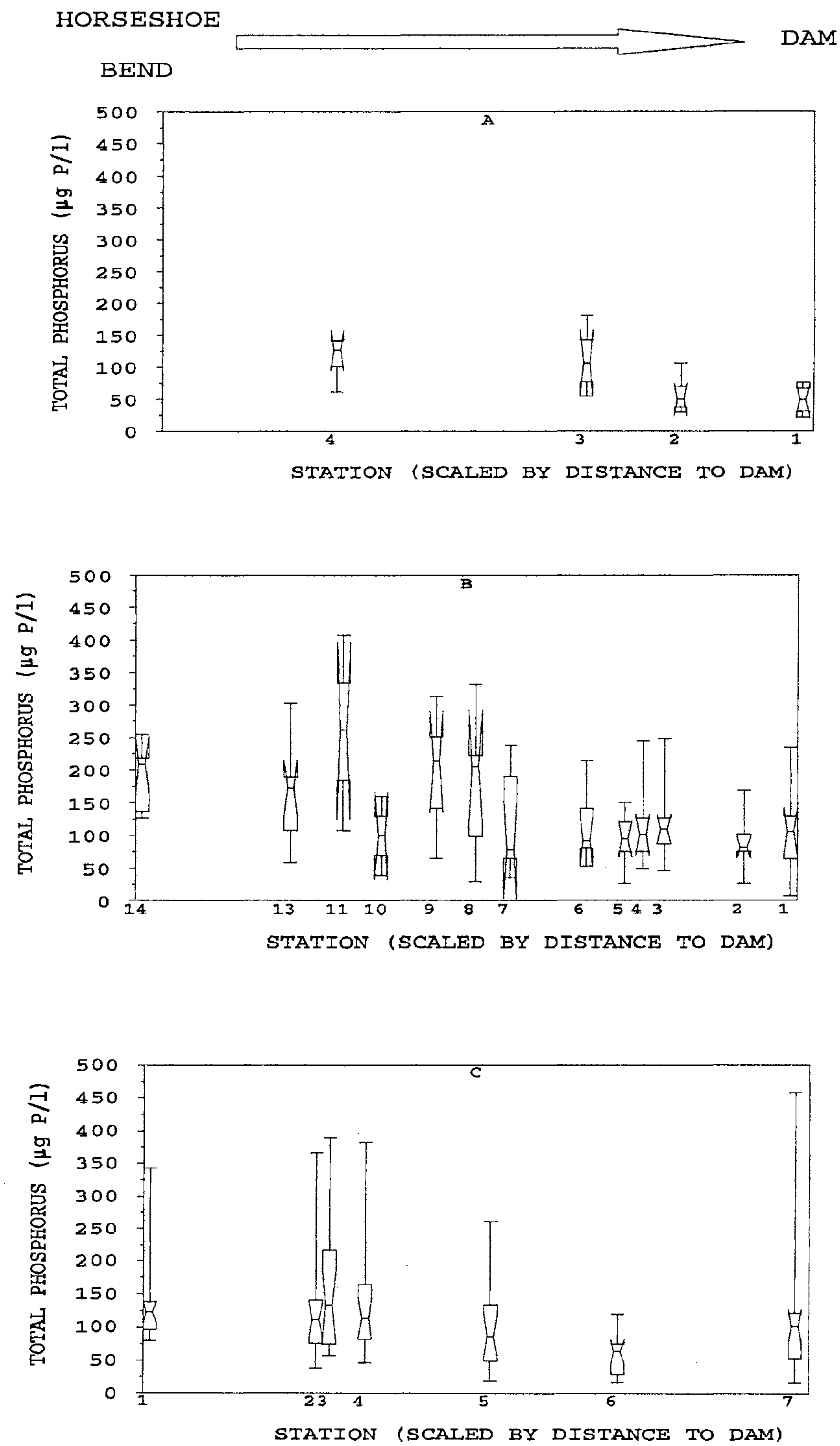

Figure 5. Hypolimnetic Total P Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

was 355 μg N/l and increased to 1000 μg N/l at the next downstream station. Following this increase, nitrate levels showed an overall decrease with a relatively high variability. The EPA-NES study in 1974 showed epilimnetic nitrate N concentrations similar to to that observed in 1992-93 (Figure 6).

The only comparisons of total N trends are between the EPA-NES 1974 study and the current CLP 1992-93 study; total N data for the USACE study were not available for comparison. Total N trends, like nitrate N, showed a similar trend in 1992-93 decreasing from a median of 2180 μg N/l at station 1 to 1160 μg N/l at station 2 (Figure 7). Median values of epilimnetic total N continued to decrease to station 5 and stabilized thereafter. These 1992-93 median values were not significantly different than those observed in the 1974 EPA-NES study. All median values observed exceeded the eutrophic criteria proposed by Sawyer (1947).

The implication of total N content in lake eutrophication is relative to the amount of phosphorus (see Sakamoto 1966). The philosophy of using the atomic ratio total N:total P was discussed earlier (see ELA project in Chapter II). The 1992-93 median total N:toal P trend observed in Lake Tenkiller exceeded threshold criteria proposed by Vollenweider (1968) and Sakamoto (1966) and thus suggests P-limitation (Figure 8). Phosphorus limitation in Lake Tenkiller also was indicated by algal assays conducted concurrently by Haraughty (personal communication).

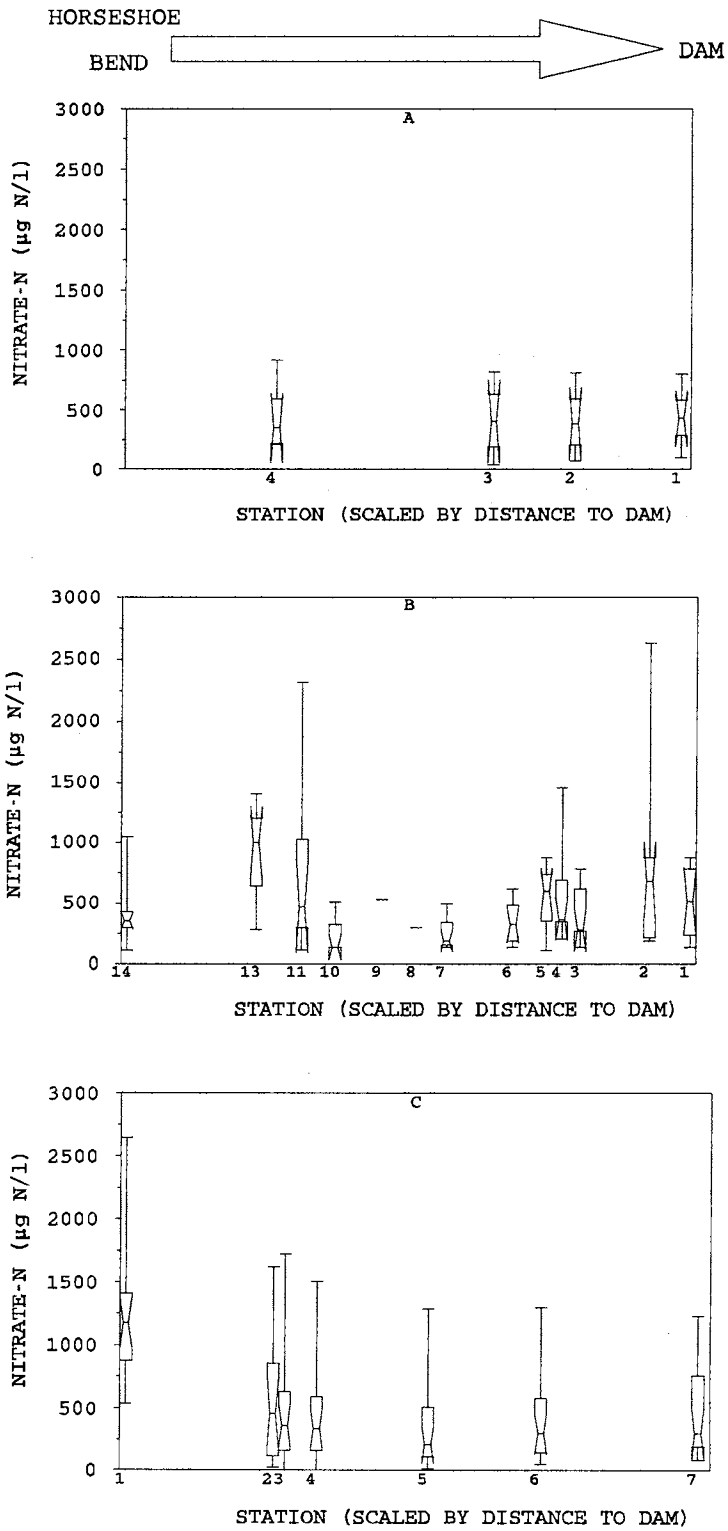

Figure 6. Epilimnetic Nitrate N Concentrations in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-1993.

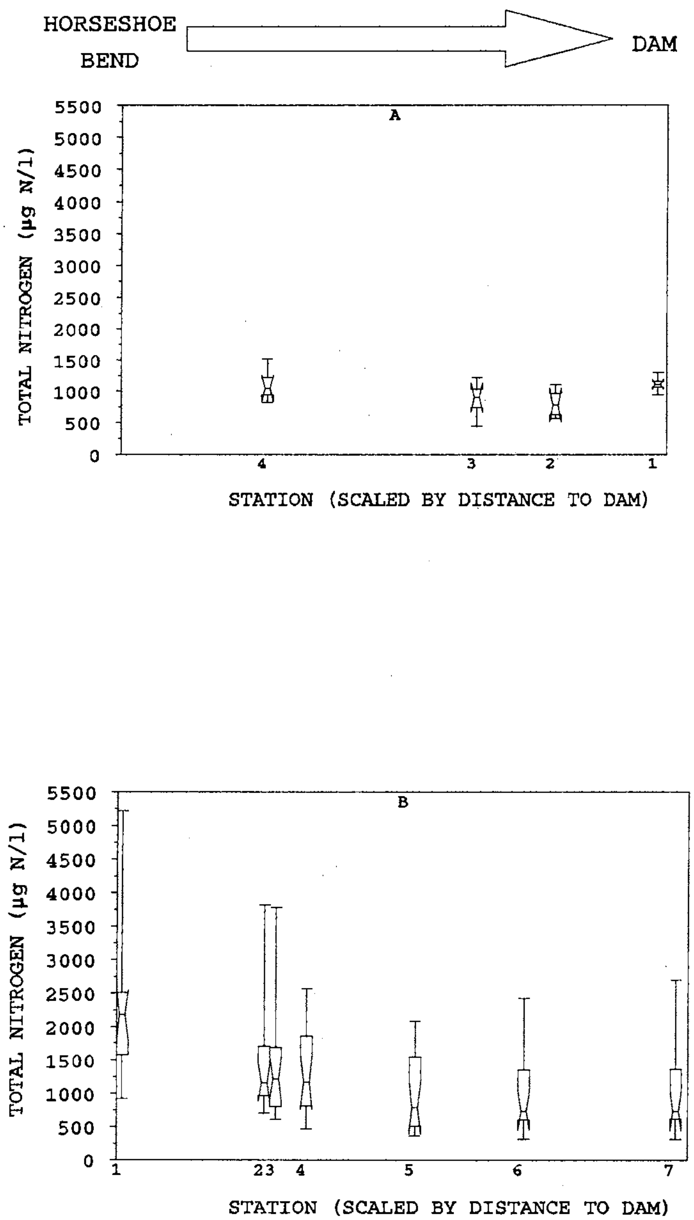

Figure 7. Epilimnetic Total N Concentrations in Lake Tenkiller for (A) EPA-NES in 1974 and (B) CLP in 1992-93.

Therefore, nitrogen-limitation was not implied, nor was it detected in these nutrient limitation assays. Close examination of Figure 8 illustrates that at times Lake Tenkiller may exhibit total N:total P ratios ≤ approximately 15 as far downstream as station 6. Therefore, the lake could exhibit undesirable conditions of blugreen dominance, etc., if P loads continue to increase and shift these distributions downwards.

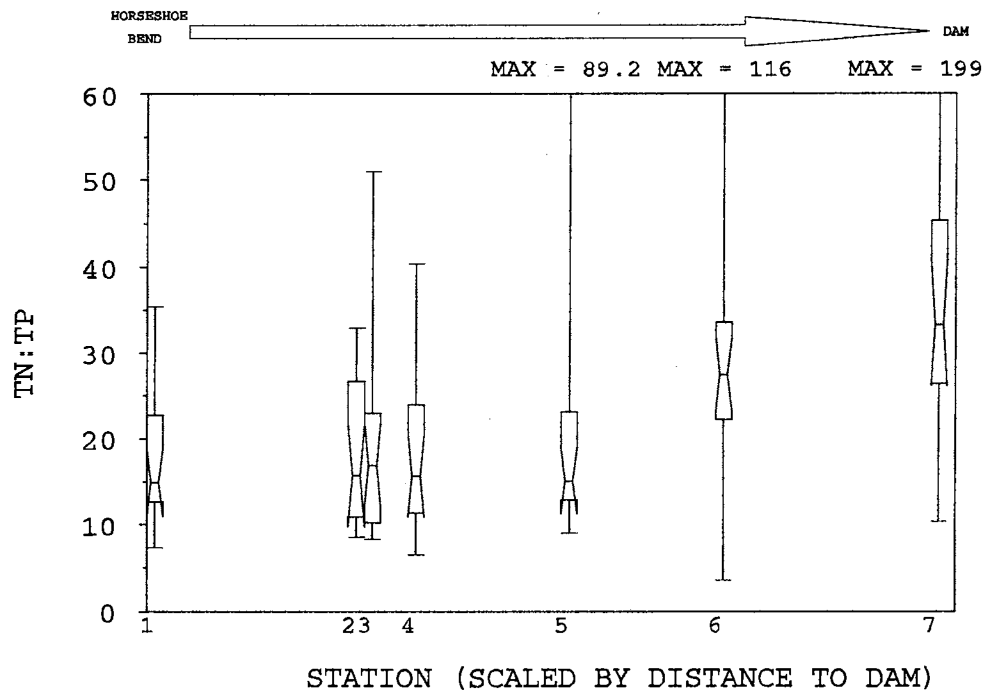

Figure 8. Total N:Total P Trends in Lake Tenkiller for CLP Study of 1992-93.

Turbidity trends were compared for USACE 1985-86 and CLP 1992-93 studies only because no comparable turbidity data for the EPA-NES 1974 study was available. The epilimnetic turbidity trend for both studies were similar. These trends exhibited a sigmoidal spatial trend in which the maximum turbidity was at Horseshoe Bend and had a maximum decay at approximately 1/3 to 1/2 the thalweg distance (i.e., CLP stations 2-4 and USACE stations 11-8) (Figure 9). Absolute epilimnetic turbidities were not significantly different between the two studies.

Hypolimnetic turbidities exhibited a strong contrast between the two studies. The spatial trend in the USACE 1985-86 study indicated a sigmoidal trend as was detected in the epilimnion while the CLP 1992-93 study showed an opposite trend in which turbidity increased with thalweg distance (Figure 10). I speculate the extreme turbidity at station 7 of the CLP study is due partially to effects of hypolimnetic withdrawal. The increase in turbidity along the thalweg at the other stations might be due to underflow. If underflow is the predominant direction of influent water, these flows coupled with hypolimnetic withdrawal conceivably could increase turbidity as it approaches the dam because hypolimnetic sediments become less consolidated with thalweg distance (these are the fine silts).

Secchi disk depths showed a spatial sigmoidal trend for all three studies where the lowest Secchi disk depths were observed in the headwaters and increased to a maximum near the dam (Figure 11). The largest increases in depth along the thalweg were slightly downstream from the largest decrease in turbidity (cf. Figures 9 and 11). No significant temporal trend was detected; in fact the median Secchi disk depths were generally larger in 1992-93 and 1985-86 than in 1974 (Figure 11). However, the variability

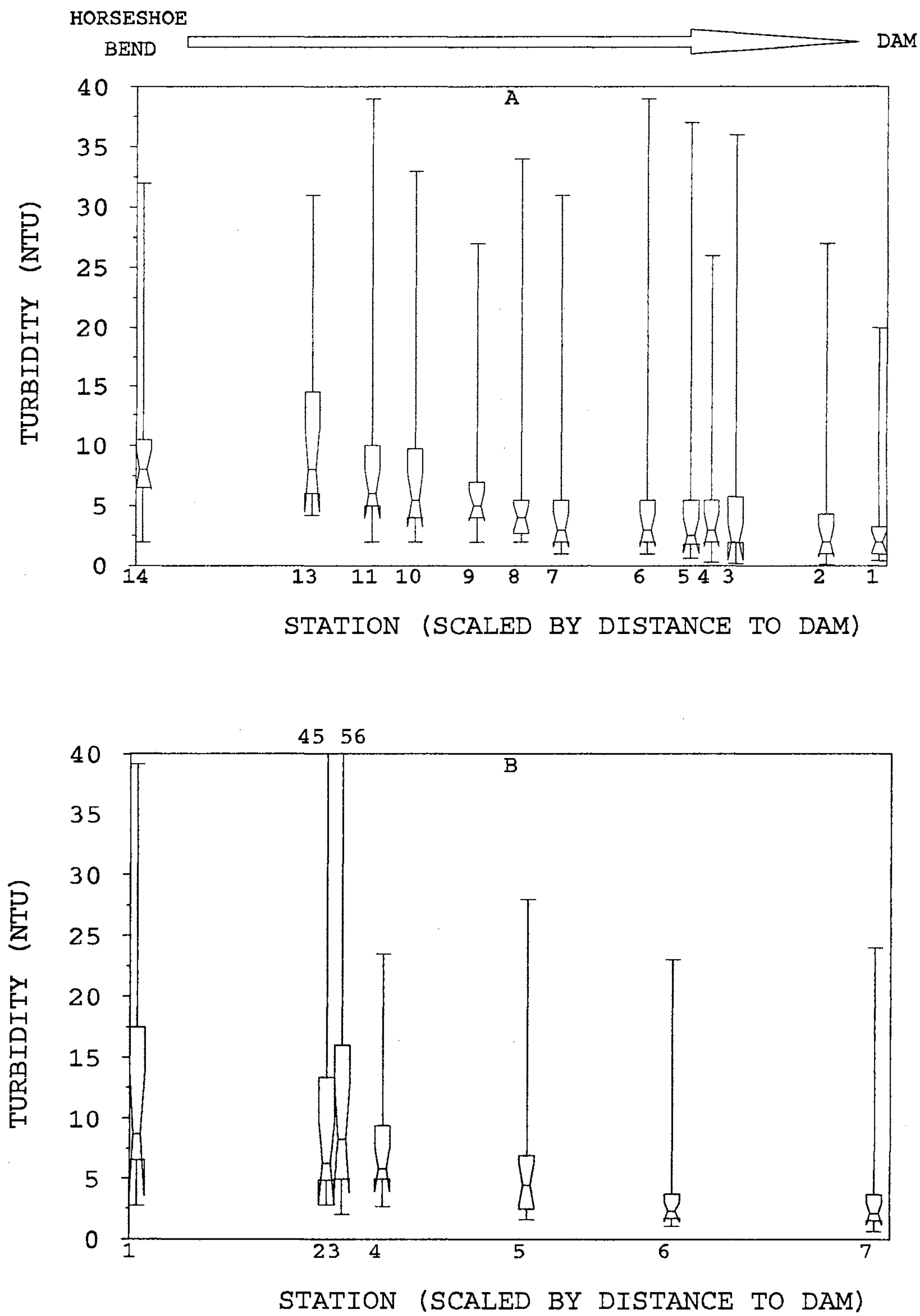

Figure 9. Epilimnetic Turbidity Trends in Lake Tenkiller for (A) USACE in 1985-86 and (B) CLP 1992-93.

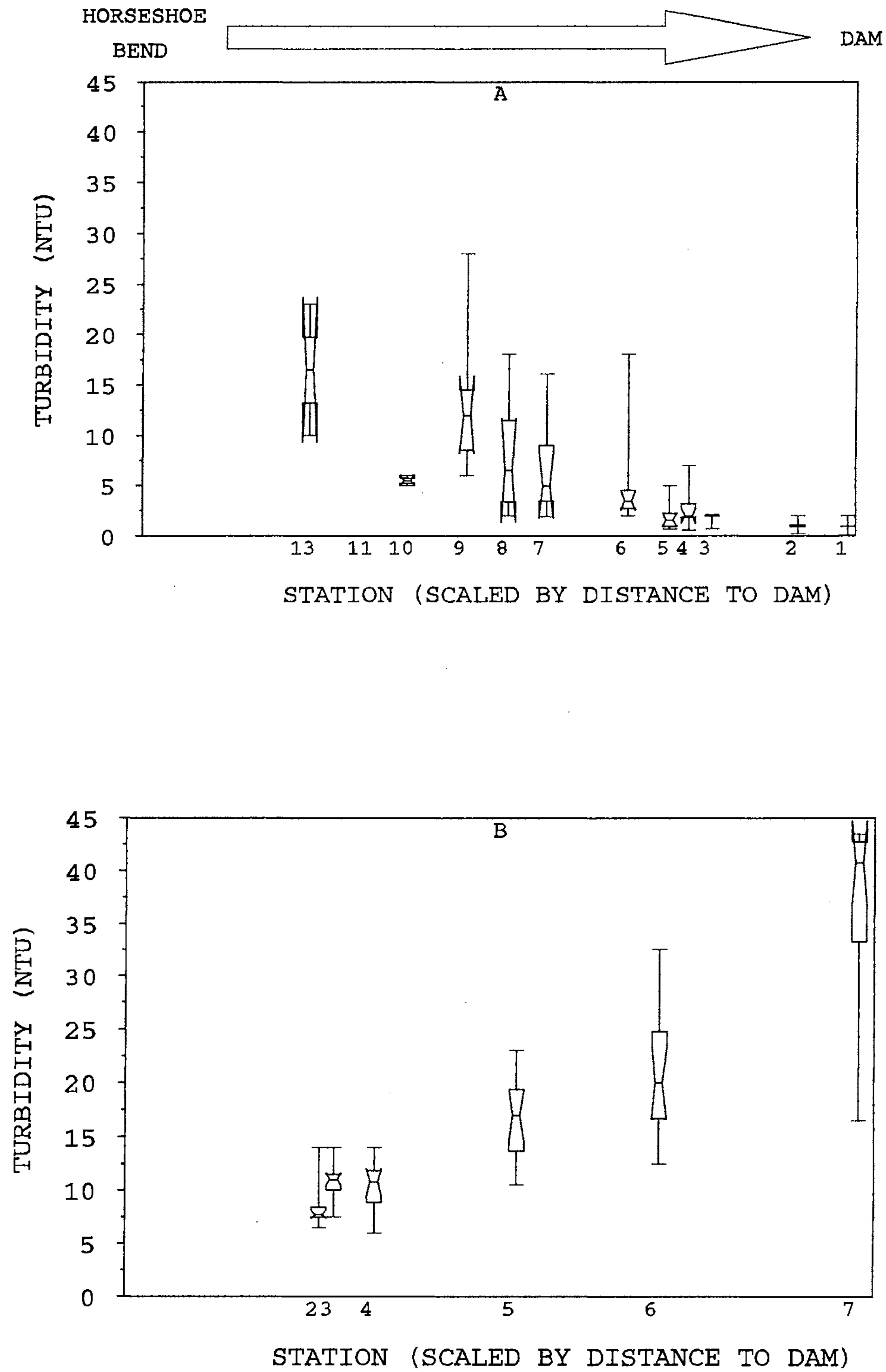

Figure 10. Hypolimnetic Turbidity in Lake Tenkiller for (A) USACE in 1985-86 and (B) CLP in 1992-93.

and the sample size of the EPA-NES 1974 study might limit the value of such generalizations.

The observed trends in Secchi disk depths given in Figure 11 are not surprising when you consider the close association of transparency (i.e., Secchi disk depth) and turbidity. Both have been evaluated here because; 1) although closely related, transparency and turbidity are not measures of the same property and 2) for reduction in model error a model parameter descriptive of light availability with less variability was desired.

In 1992-93, chlorophyll *a* exhibited a maxima relationship with a median concentration of 28.6 μg/l at station 2. Because station 2 was located in the mouth of Caney Creek (Figure 1), the peak may have been due to a "cove effect" of the tributary and thus did not represent a main thalweg effect. However, Station 4 was in the mainstem thalweg (Figure 1) and had a median chlorophyll *a* of 28.7 μg/l which was not significantly different than station 2 or 3 (Figure 12). Station 1 showed the lowest median chlorophyll *a* density of all stations (excluding station 8, the tailrace) with a median density of 2.5 μg/l (Figure 12).

In 1985-86, chlorophyll *a* trends were similar to that observed in 1992-93 where comparable stations were slightly higher in 1985-86 but the differences were not statistically significant (Figure 12).

A statistically significant increase in chlorophyll *a* density at the point of maximum density and near the dam has occurred in Lake Tenkiller since the 1974 EPA-NES study

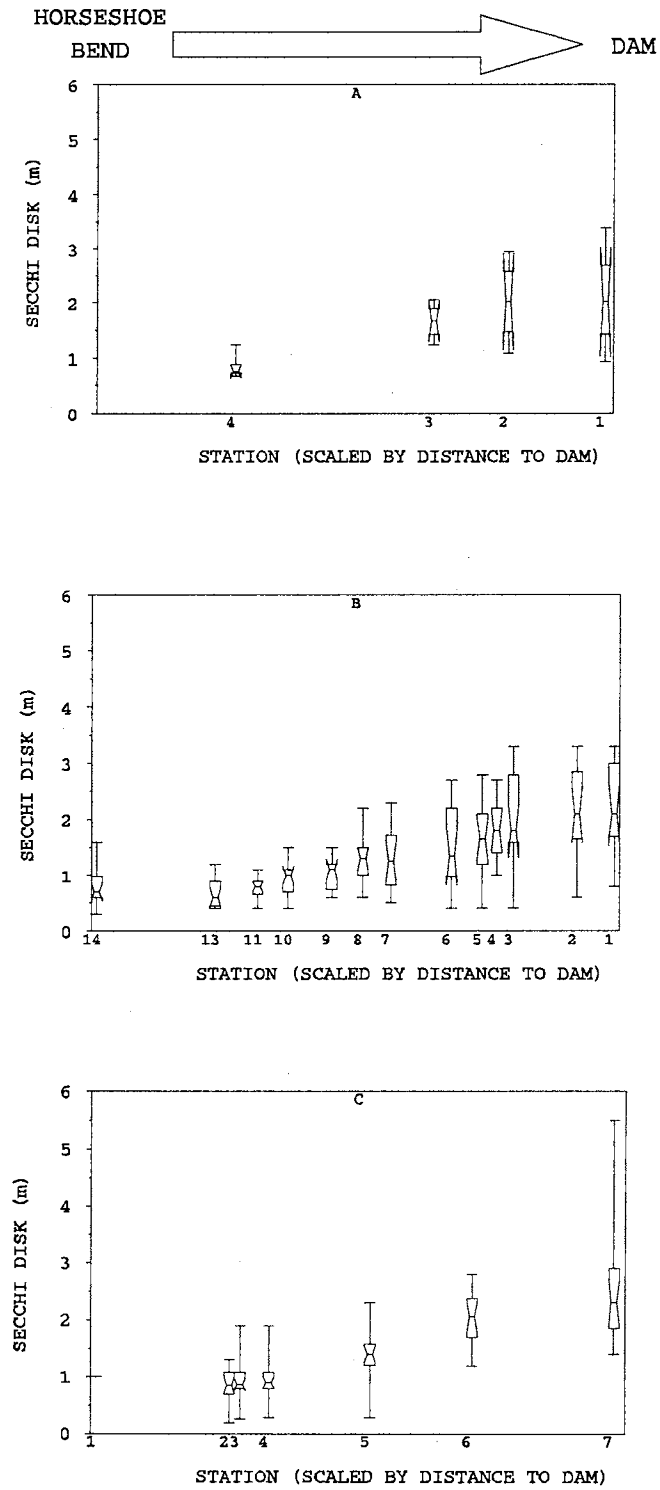

Figure 11. Secchi Disk Depths in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

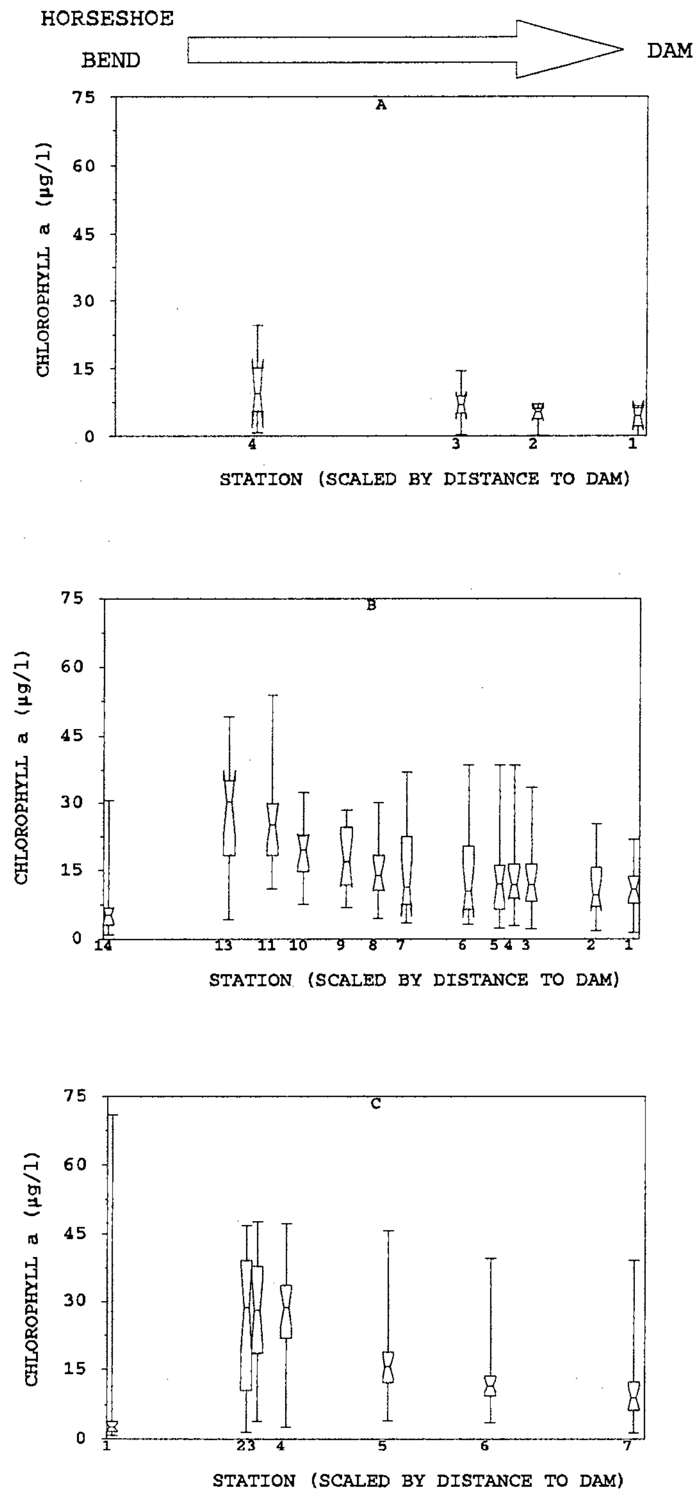

Figure 12. Chlorophyll *a* Densities in Lake Tenkiller for (A) EPA-NES in 1974, (B) USACE in 1985-86, and (C) CLP in 1992-93.

(Figure 12). Interestingly, an increase in variance of the chlorophyll densities also has occurred (Figure 12). I believe this is due to the increased amplitude of total P loads associated with runoff (cf. Figures 5 and 12). Also, a characteristic of increasing stress is an increased variance in biotic structure (Barrett et al. 1976, Odum 1975, Odum 1979).

In summary, I conclude that Lake TenKiller is in the advanced stages of eutrophication because:

1) total P loads have significantly increased since 1974 while corresponding increases in N fractions were not detected, and

2) chlorophyll *a* densities along the thalweg have increased significantly since 1974.

Evaluating this trend quantitatively and incorporating an element of risk follows with construction of the hydraulic budget and RERF model development.

Hydraulic Budget

An empirical analysis indicated that the capacity and area could be estimated from lake elevation. The empirical relationship derived was based upon data provided by USACE. Nonlinear regression for capacity on elevation yielded:

$$CAPACITY = 197.2 * e^{0.135 * ELEVATION} - 527.8 * ELEVATION \qquad \textbf{(29)}$$

where CAPACITY is in ac-ft and ELEVATION is ft msl. Area was regressed on elevation and yielded:

$$AREA = 2.195 * e^{0.0137 * ELEVATION} \quad (30)$$

where AREA is in acres and ELEVATION is ft msl.

Hydraulic residence time varied from 0.033 to 3.4 yr with a mean of 0.76 yr (Figure 13). The maximum residence times were observed during ca. Aug - Sep 92 and Aug 93, while the minimum residence times were observed in early 1993 (Figure 13). Undoubtedly, the observed trend was due to seasonality of runoff and the 1993 spring flood. The implication of the variability in residence time (more than trebled) to the development of a longitudinal transport model (RERF) is that decreased residence time indicates increased flow through the reservoir and thus would be expected to "push" the transition zone towards the dam.

The daily hydraulic balance approximated equilibrium conditions except for ca. May - Jun 92 and ca. Dec 92 - Jun 93 (Figure 14). An individual peak in Figure 14 did not necessarily indicate the reservoir was in a state of disequilibrium because dam discharge schedules may dictate no flow downstream for 24 hr. The breadth of the peaks in Figure 14 is more indicative of the reservoir as being "filled" or "emptied". However, comparison of Figures 14 and 15 indicated that the four largest peaks of inbalance (Figure 14) immediately preceded the largest increases in lake elevation. This trend is discussed further in evaluation of the RERF model.

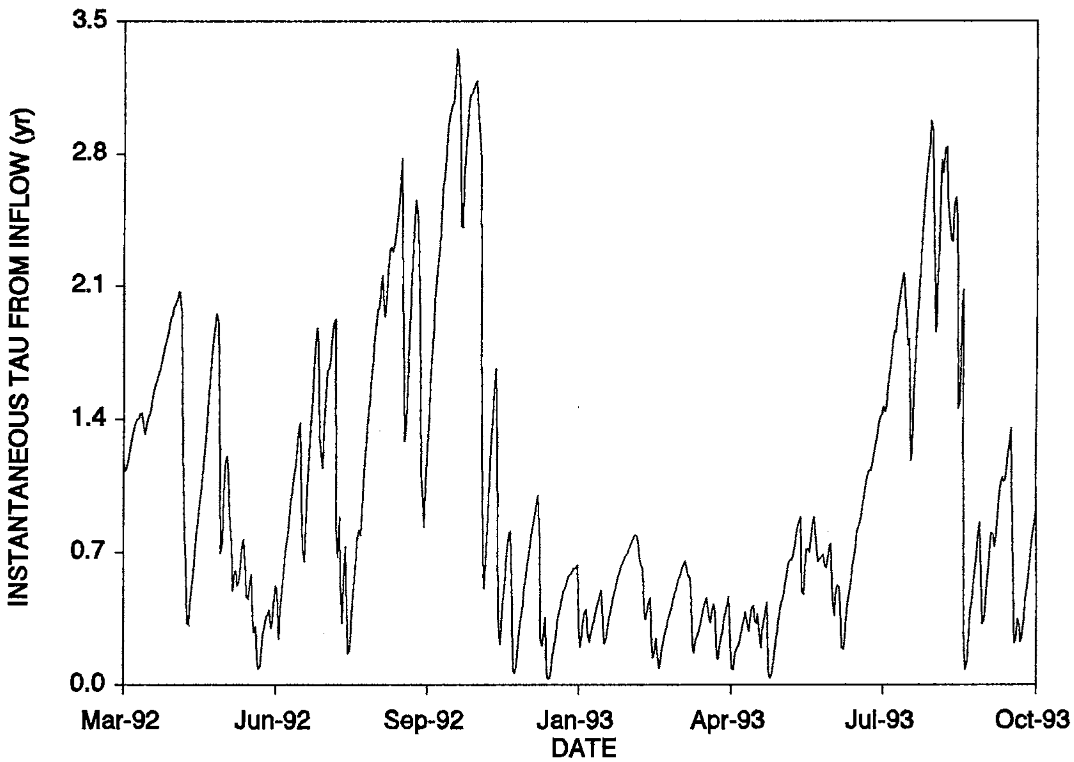

Figure 13. Daily Hydraulic Residence Time for Lake Tenkiller, CY 92-93.

Reservoir Ecological Risk Factor Model

<u>Nonlinear Regression Results</u>

The TP sigmoid equations fit to the data collected during the CLP project (CY 1992-93) yielded an EL50 of 29% (95% confidence interval = 21 to 38%) of the thalweg distance, a slope factor of -7.04 (95% confidence interval = -11.8 to -2.3), and an estimated MIN of 33 μg P/l (95% confidence interval = 16.3 to 49.8) (Figure 16). The variance in measured TP decreases nearer the dam (Figure 16). The smaller model variance in the upper end is an artifact of setting the median TP at station 1 as the

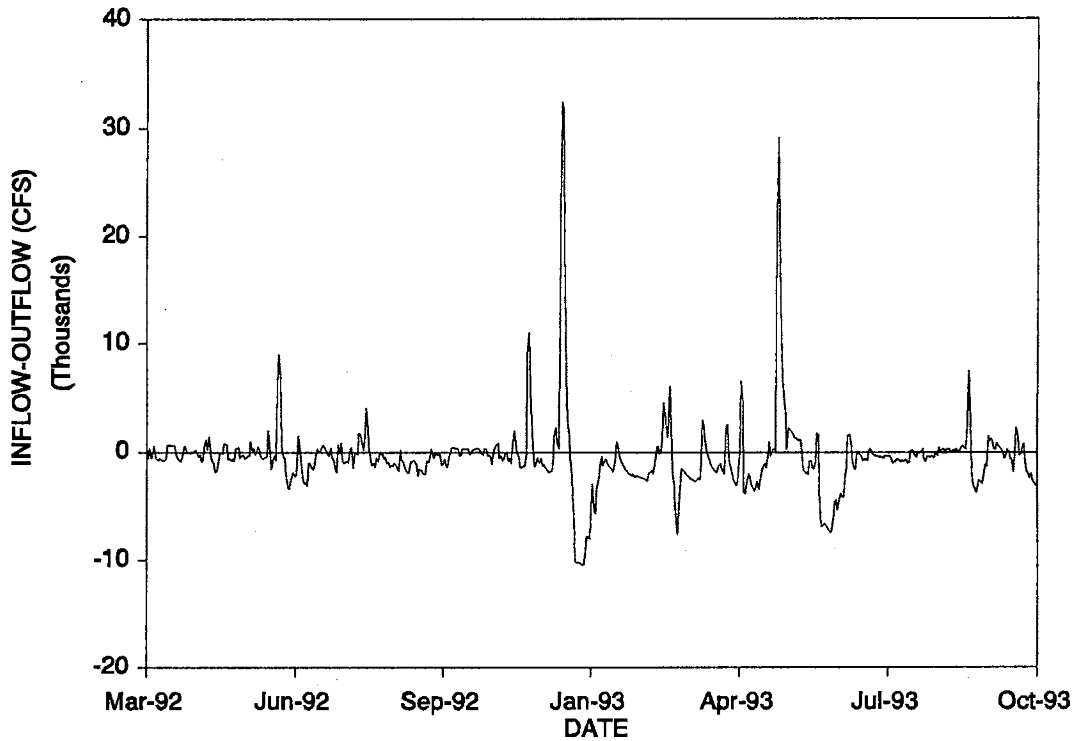

Figure 14. Hydraulic Balance of Lake Tenkiller for CY 92-93.

absolute MAX in the equation. This constraint was necessary because initial best fit equations estimated a MAX TP much greater than any observed in the lake or in the immediately-upstream gauging stations (USGS07196500 or USGS07197000) and thus was considered to be unrealistic.

Nonlinear regression results for the USACE 1985-86 data yielded a MIN = 93.3 μg P/l (95% confidence interval = 65.8 to 120), a slope factor = - 5.339 (95% confidence interval = -9.78 to -0.893), and an EL50 = 35.7% of the thalweg distance (95% confidence interval = 20.8 to 50.6) (Figure 16). The higher minimum estimated for the 1985-86 data indicated a decrease in total P at the lower end of the lake by 1992-93. However, the model maximum for 1985-86 data was

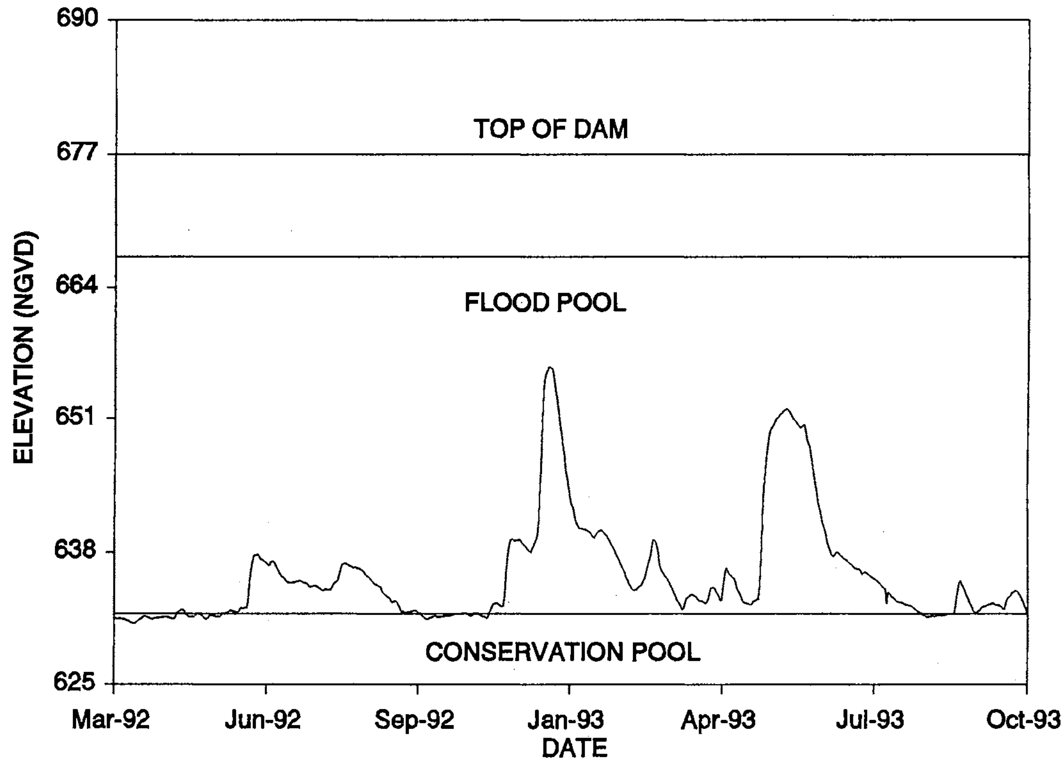

Figure 15. Daily Elevations (NGVD) of Lake Tenkiller for CY 92-93 (24 hr means of hourly data provided by USACE).

set at 211 µg P/l (median at Horseshoe Bend) which is almost twice that during 1992-93 (130 µg P/l). The relationship illustrates that total P levels near the dam will respond to decreases in headwater loads. This is an important result for recommended management in Lake Tenkiller.

Turbidity data from the USACE 1985-86 study fit to the sigmoidal equation yielded a MIN = 2.223 nephelometric turbidity units or NTU (95% confidence interval = 11.9 to 0 NTU), a slope factor of -8.438 (95% confidence interval = -23.9 to -7.0), and an EL50 = 80% (95% confidence interval = 29.3 to >100%) of the thalweg distance. The MAX was set to 8 NTU (the median of the headwater station). These best fit results given in Figure 17 - panel A appear to overestimate

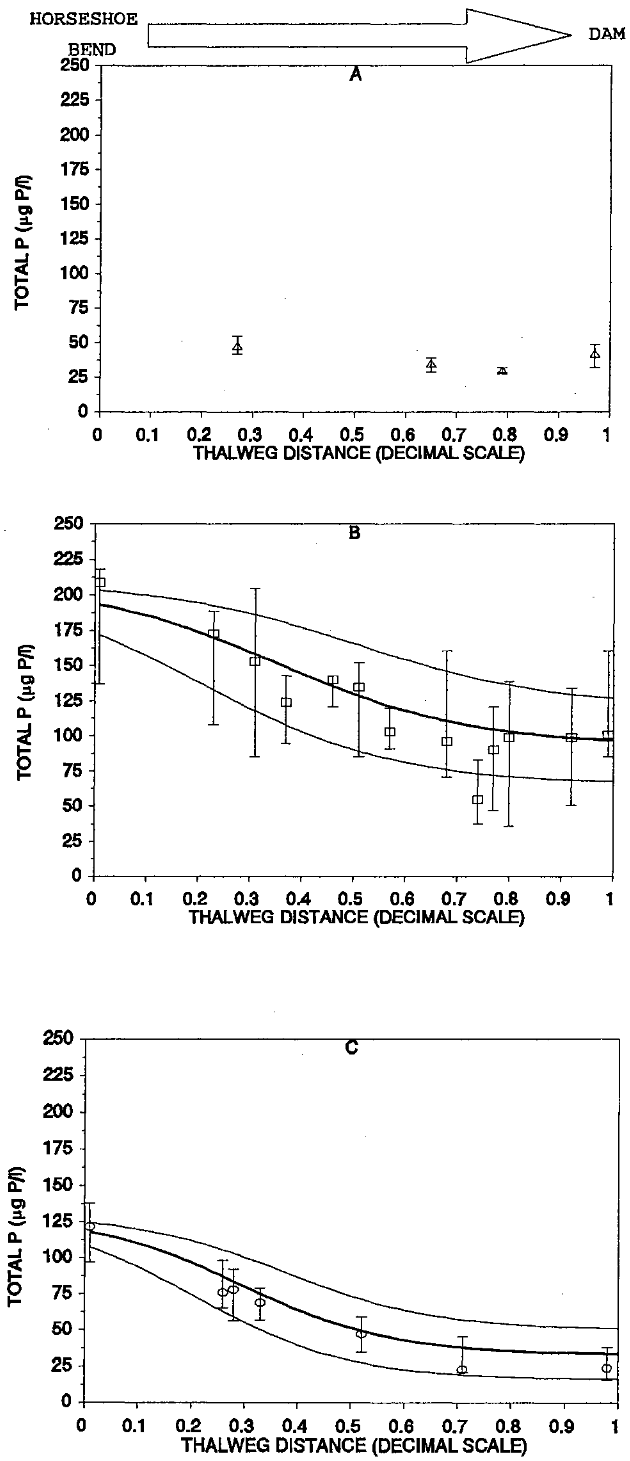

Figure 16. Nonlinear Regression Results for Epilimnetic Total P; (A) 1974 (data only), (B) 1985-86, (C) 1992-93 (lines=±2 SE of model; error bars=quartiles).

the true trend indicated by the error bars (quartiles). However, comparison of Figure 17 - panel A with Figure 9 illustrates that extreme outliers (maximum turbidities) during the USACE study biased the estimates of the regression. The RERF is based on chlorophyll *a* trends and not quantitative turbidity; thus data "culling" was not performed.

Turbidity data for the CLP 1992-93 study fit to the sigmoidal equation yielded an EL50 of 33% (95% confidence interval = 29 to 37%), a slope factor of -122 (95% confidence interval = -1260 to 1020), and a MIN of 4.73 nephelometric turbidity units or NTU (95% confidence interval = 1.99 to 7.47 NTU; Figure 17). The slope factor's confidence interval may seem wide but, considering the parameter approaches infinity as the slope approaches a vertical line, the estimate is realistic. Therefore, the high slope factor merely indicates a rapid decline in turbidity near the EL50 (Figure 17). For this reason, the error estimates of the slope factors were not included in the sigmoidal graphs.

Overall, the trend in turbidity has not changed significantly since the USACE 1985-86 study. Both studies illustrated higher turbidities and variances in the headwaters giving way to a lower and more stable turbidity near the dam. The only turbidity assumption the RERF requires is a sufficient decay so that light availability changes from limitation to saturation.

Secchi disk data from the EPA-NES 1974 study yielded a

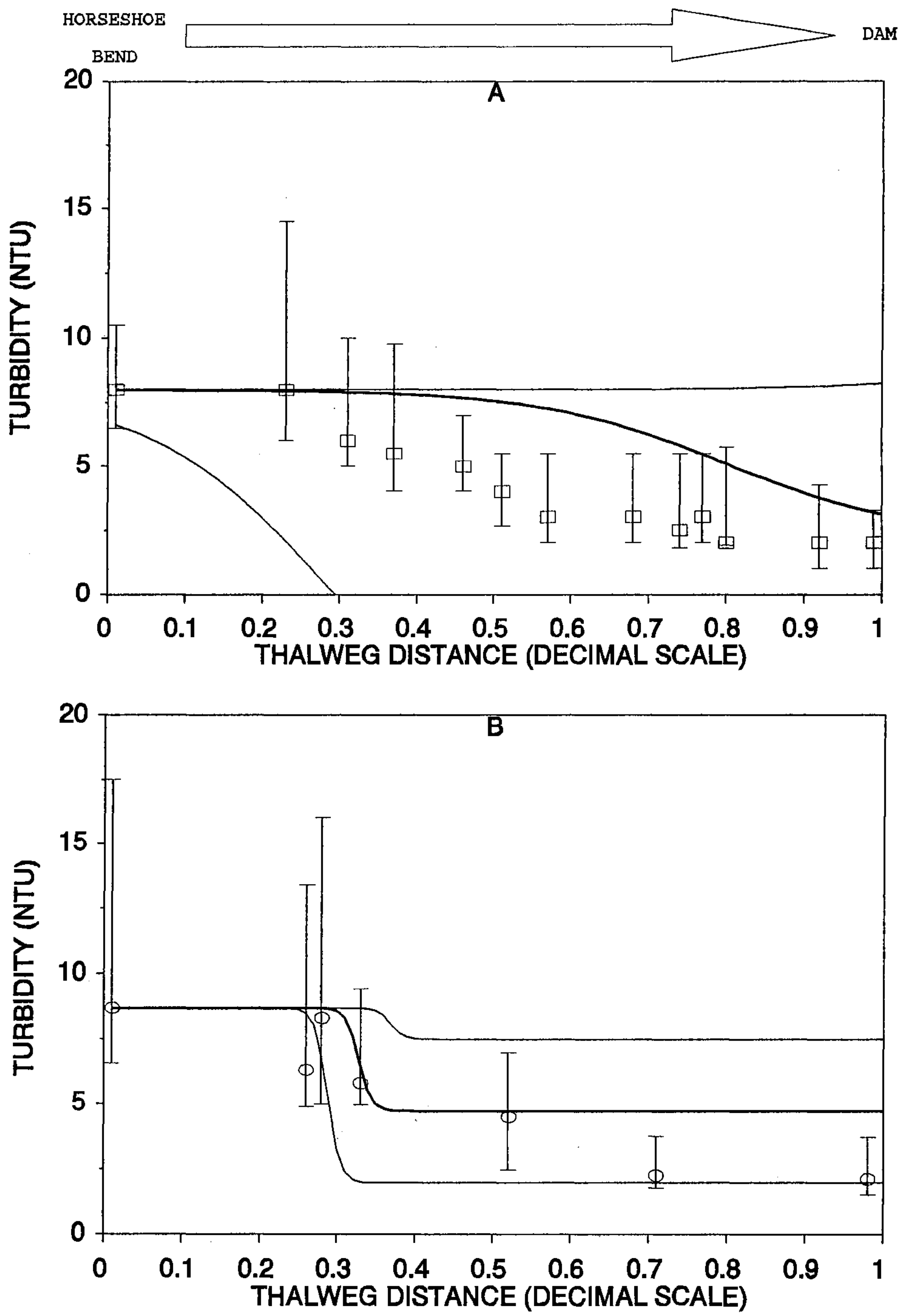

Figure 17. Nonlinear Regression Results for Turbidity; (A) 1985-86, (B) 1992-93; (lines=±2 SEM of model; error bars=quartiles).

MAX = 21.4 dm (95% confidence interval = 12.5 to 30.3 dm), a slope factor = 9.56 (95% confidence interval = -22.9 to 42), and an EL50 = 58% (95% confidence interval = 23.9 to 92.8%) of the thalweg distance (Figure 18). The USACE 1985-86 data yielded a MAX 24.9 dm (95% confidence interval = 18.9 to 31.0 dm), a slope factor = 6.714 (95% confidence interval = 3.57 to 9.86), and an EL50 = 68.7% (95% confidence interval = 55.2 to 82.3%) of the thalweg distance. The Secchi disk data for the CLP 1992-93 study yielded an EL50 at 63% (95% confidence interval = 54 to 71%), a slope factor of 9.13 (95% confidence interval = 4.61 to 13.6), and an estimated MAX of 25.8 dm (95% confidence interval = 22.3 to 29.3) (Figure 18). Again, decimeters were used in lieu of the standard meter to approximate more closely the range (absolute) of TP, turbidity, and chlorophyll.

Also, Secchi disk EL50s were greater than turbidity EL50s and thus implied the former was more sensitive at the lower end of the lake (i.e., higher transparencies). This is plausible because turbidity measures scattering of light while the Secchi disk measures transparency or penetration depths. The increased variance of the Secchi disk nearer the dam (Figure 18) and vice versa for turbidity (Figure 17) tends to support this conclusion.

Overall comparison of the trends illustrated in Figures 17 and 18 indicated no significant change in turbidity since 1985-86 or Secchi disk since 1974. This is antithetical to traditional concepts of accelerated eutrophication which leads to increased turbidity and decreased transparency.

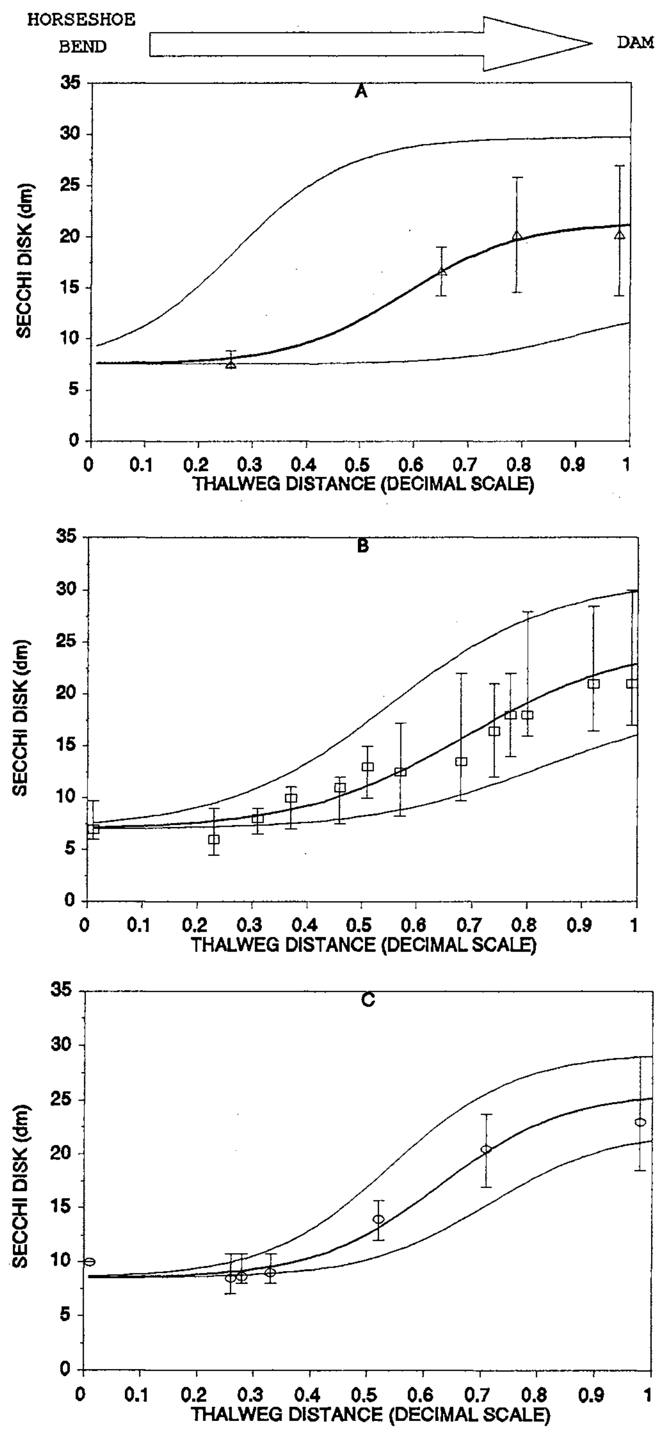

Figure 18. Nonlinear Regression Results for Secchi Disk; (A) 1974, (B) 1985-86, (C) 1992-93 (lines=±2 SE of model; error bars=quartiles).

Hence, I propose chlorophyll *a* is the primary response variable to be modelled as an indicator of trophic status, and thus temporal trends in turbidity are irrelevant assuming the transition from light limitation to saturation.

Chlorophyll *a* data from the EPA-NES 1974 study fit to the maxima function resulted in a CRF = 105 (95% confidence interval = 7.13 to 203) and a PF = -3.50 (95% confidence interval -5.29 to -1.72). The USACE 1985-86 study resulted in a CRF = 209 (95% confidence interval = 154 to 263) and a PF = -3.32 (95% confidence interval = -3.82 to -2.82). The CLP 1992-93 study yielded a CRF = 254 (95% confidence interval = 177 to 331) and a PF = -3.48 (95% confidence interval = -4.22 to -2.73). Unlike the TP, turbidity, and Secchi disk depth models, initial chlorophyll *a* value is set as 0 μg/l as a model constraint (a default for the maxima function). While this is obviously untrue, the constraint is irrelevant to model interpretation because impacts of nutrients do not occur until the transition zone and beyond.

These coefficients indicated a statistically significant increase in chlorophyll density at the maxima (i.e., transition zone) from 1974 to 1985-86 and no significant change since 1985-86 (Figure 19). Model results shown in Figure 19 indicated the largest variation was in the transition zone which graded to a more stable population downstream and upstream from the maximum occurrence (peak). Also, the feature in comparing panels A, B, and C in Figure 19 is an obvious temporal increase in variance in chlorophyll *a* densities at the peaks. This is a classic sign of a "stressed" ecosystem (Odum 1975, 1979).

In summary, the nonlinear regression efforts resulted in the following conclusions:

1) no statistically significant temporal changes in turbidity or Secchi disk transparencies could be detected, however, spatial trends were statistically detected;

2) a statistically significant (as per McGill et al. 1978) temporal increase in total P levels was detected at headwaters and transition zones; and

3) a statistically significant temporal increase in chlorophyll *a* densities at the transition zone was detected.

<u>Maxima Function Interpretation</u>

For each sampling date during the CLP 1992-93 and USACE 1985-86 studies, the nonlinear fit was performed and resulting parameters evaluated. Lake Tenkiller exhibited accelerated eutrophication (beyond mesotrophy as delineated by chlorophyll *a* densities > 10 μg/l) on all dates (Figure 20). Lake Tenkiller was not definitively classified according to trophic status. The boundaries of mesotrophy, eutrophy, hypereutrophy, etc. were not delineated (Figures 3 or 20). I conclude the reservoir was in advanced stages of eutrophication the further the points lie to the upper right of the figure (i.e., higher CRF and PF). Two sampling dates for the CLP study are not shown in Figure 20, because they were off the scale. On 18 Apr 93, the model yielded CRF = 5.19 and PF = 1.80, and on 26 May 93, the CRF estimate was 8070 and PF = -14.7. The 18 Apr 93 results reflected a chlorophyll *a* trend where the maximum was at station 7 (31

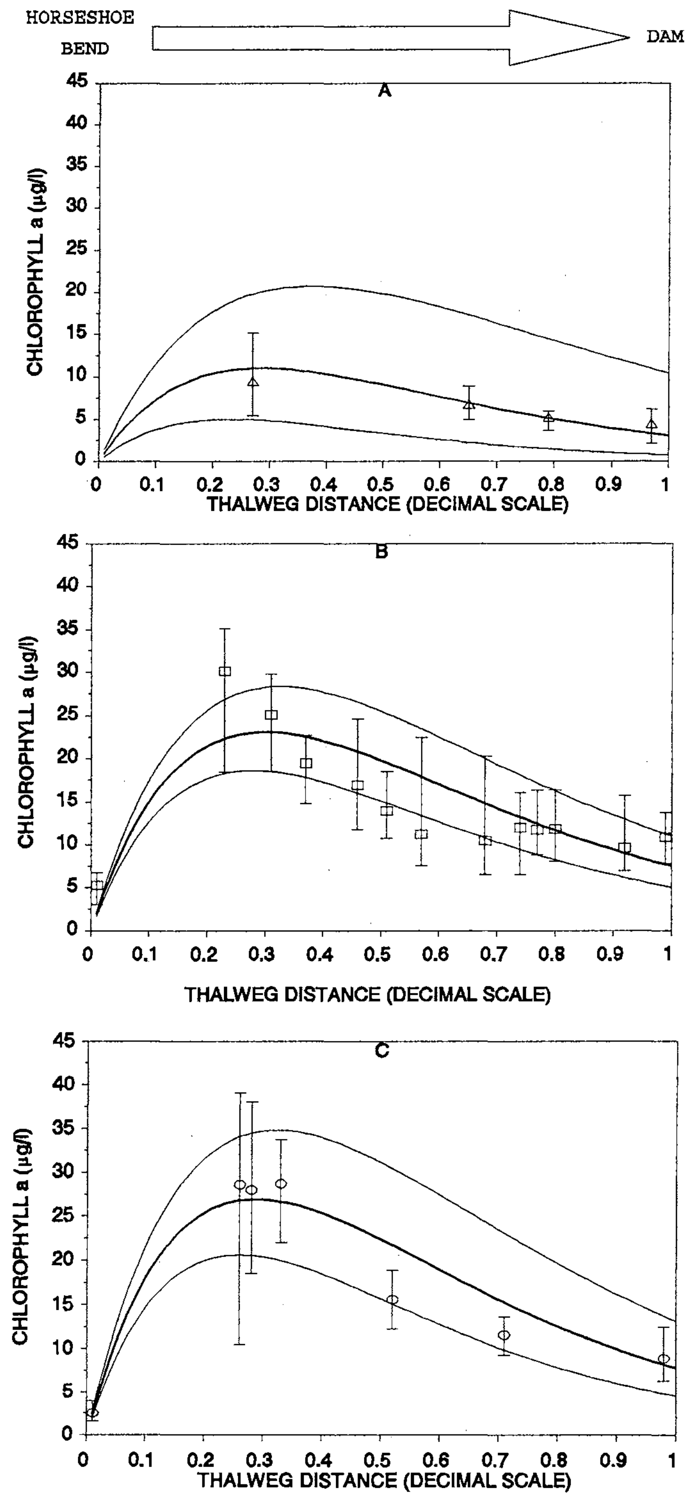

Figure 19. Nonlinear Regression Results for Chlorophyll *a*; (A) 1974, (B) 1985-86, (C) 1992-93 (lines=±2 SE of model; error bars=quartiles).

μg/l) while the upstream stations were 11 μg/l or less. As an indication of how fast the reservoir conditions can change, the other outlier was on 26 May 93 when the opposite was observed. On this date, station 1 showed 71 μg/l while the remaining stations steadily decreased to approximately 8 μg/l at station 7 (Appendix A).

Interpretation of the confidence intervals of the model parameters also indicated a significant trend of accelerated eutrophication in Lake Tenkiller since 1974 (Figure 20).

Model Formulation

One of my hypotheses was that hydraulic phenomena affected the impact of the nutrient load. Therefore, to gain insight on these effects, I constructed a correlation matrix of model parameters on hydraulic conditions during the CLP 1992-93 study (Table VII). Initial evaluation of the matrices indicated that the capacity, z_{avg}, and area were closely correlated with CRF. Actually, this is misleading. Area and capacity were calculated from the same datum, elevation and z_{avg} is capacity/area. Therefore, any correlation of these to CRF PF can only be inferred from one of these morphometrics.

The peak height (CAMAX) was significantly correlated with mean depth, while the distance at which the peak occurred was better correlated with the hydraulic balance and z_{avg}/τ_w, albeit not significantly (cf. Tables VII and VIII). Peak height was also negatively correlated with the

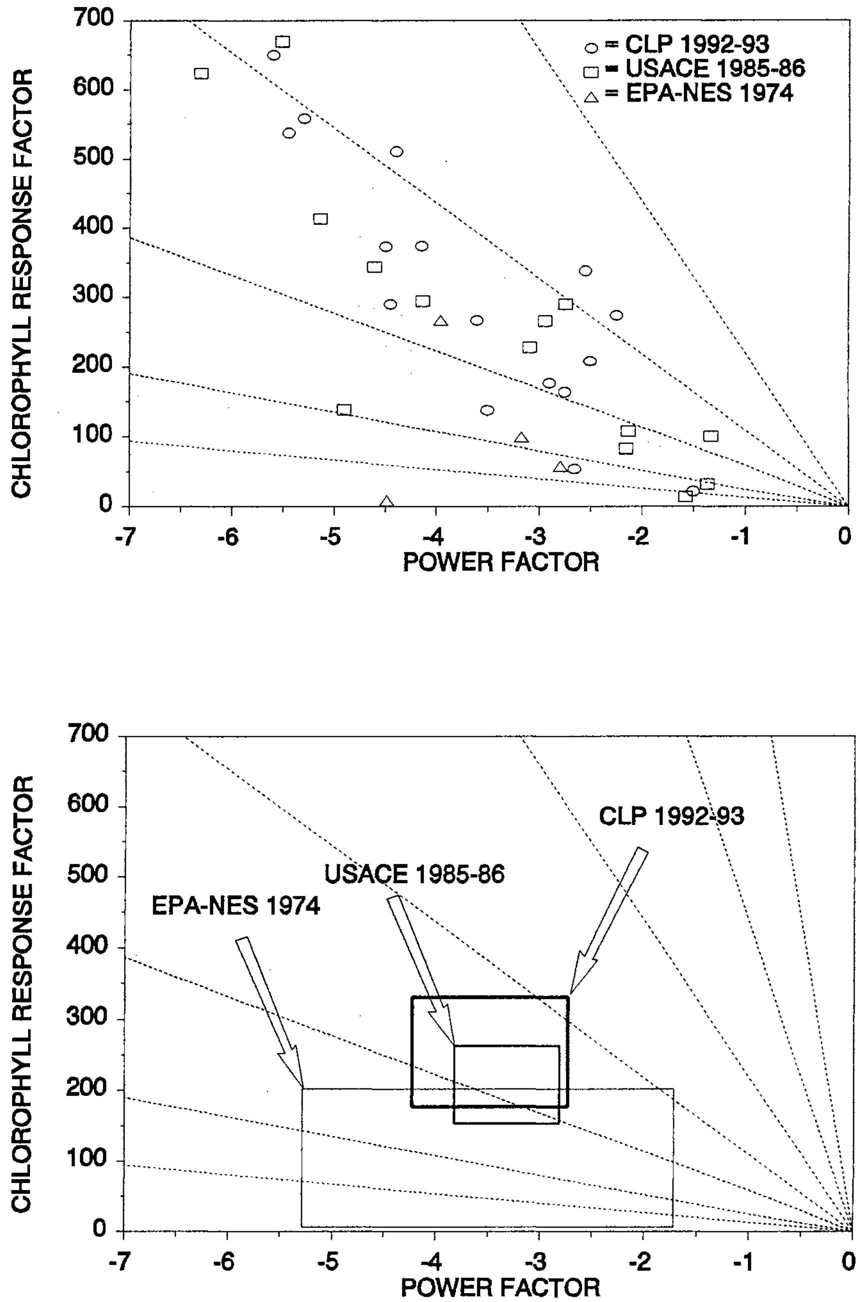

Figure 20. Integration of Maxima Model Coefficients in Lake Tenkiller for Each Sampling Event (upper) and 95% Confidence Intervals of Model Parameters (lower).

model parameter TPMAXRED ($r = -0.49$, $p < 0.05$) and positively correlated with turbidity minimum ($r = 0.62$, $p < 0.05$). This relationship is plausible because as TP reduction is hampered and turbidity is minimized, algal growth would be expected to increase. The RERF model is based upon this relationship. The significant correlation between the turbidity maximum and L_P (areal P loading) exemplifies the close association between runoff (which translate to high P loads) and increased headwater turbidity. However, turbidity minima were not correlated with L_P and only weakly correlated with z_{avg}.

The correlation and probability matrices indicated the CRF could be predicted from z_{avg} and PF can be predicted from the CRF. However, as described by Thornton, et al. (1990) the flow regime dictates the occurrence of the longitudinal trend when inter-reservoir comparisons are made. Therefore,I performed a stepwise multiple regression analysis of CRF versus the hydraulic parameters in the correlation matrix. The result of the regression indicated that most of the variance in CRF could be explained by a combination of τ_w, capacity, hydraulic balance, and inflow/z_{avg}. The resulting equation was:

$$\ln(CRF) = -1.08 + 0.352(\pm 0.188)\tau_w + 8.50(\pm 1.83)CAPACITY + 0.206(\pm 0.029)BALANCE - 0.243(\pm 0.0414)\frac{Q_{INFLOW}}{Z_{AVG}} \tag{31}$$

where τ_w is hydraulic residence time in years, CAPACITY is km^3, balance is m^3/s, and Q_{inflow}/z_{avg} is m^2/s ($r^2 = 0.72$).

Table VII. Pearson Correlation Coefficients for Hydraulic and RERF Model Parameters (CAP=capacity; other parameters as identified in the text).

	TAU (YR)	Q_{in} (CFS)	CAP (ACFT)	ZAVG	BAL (CFS)	AREA (M^2)	$Z_{AVG}\div$ TAU	$Q_{IN}\div$ Z_{AVG}	AREA$\div$ Q_{IN}	L(P) $g/m^2/YR$
CRF	-0.22	-0.07	0.92	0.89	0.08	0.92	0.32	-0.09	-0.09	-0.12
PF	0.03	0.28	-0.68	-0.64	-0.30	-0.98	0.03	0.29	-0.08	0.37
CAMAX	-0.16	-0.10	0.90	0.87	0.06	0.90	0.27	-0.11	-0.08	-0.11
MAXD	-0.32	0.41	-0.13	-0.08	-0.55	-0.13	0.58	0.41	-0.28	0.54
TPMAX	0.07	0.53	0.02	0.05	0.08	0.02	0.14	0.53	-0.41	0.81
TPMIN	-0.26	0.02	0.41	0.43	-0.58	0.41	0.43	0.01	-0.02	0.18
TPRED	0.11	0.35	-0.49	-0.50	0.39	-0.49	-0.14	0.36	-0.25	0.31
TPEL50	0.04	-0.02	-0.25	0.25	-0.04	-0.25	-0.33	-0.02	0.21	-0.17
TBMAX	NA	0.81	0.28	0.29	NA	NA	NA	NA	NA	0.82
TBMIN	NA	-0.07	0.76	0.76	NA	NA	NA	NA	NA	-0.11
TBEL50	NA	-0.25	-0.32	-0.34	NA	NA	NA	NA	NA	-0.24
SDMAX	NA	-0.15	-0.14	-0.17	-0.09	-0.14	0.02	-0.14	0.29	-0.18
SDEL50	NA	-0.18	-0.28	-0.33	0.13	-0.28	-0.28	-0.17	0.36	-0.27

NA = NOT ANALYZED; TPRED = TOTAL P REDUCTION i.e., (TPMAX-TPMIN)/TPMAX;
MAXD = DECIMAL DISTANCE TO CHLOROPHYLL PEAK

Table VIII. Bonferroni Adjusted Probabilities for Correlation Coefficients Given in Table VII.

	TAU (YR)	Q_{IN} (CFS)	CAP (ACFT)	Z_{AVG} (m)	BAL (CFS)	AREA (m^2)	ZAVG÷ TAU	Q_{IN}÷ Z_{AVG}	AREA÷ Q_{IN}	L(P) $g/m^2/yr$
CRF	1.00	1.00	<0.01	<0.01	1.00	<0.01	1.00	1.00	1.00	1.00
PF	1.00	1.00	0.16	0.42	1.00	0.16	1.00	1.00	1.00	1.00
CAMAX	1.00	1.00	<0.01	<0.01	1.00	<0.01	1.00	1.00	1.00	1.00
MAXD	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
TPMAX	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	<0.01
TPMIN	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
TPRED	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
TPEL50	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
TBMAX	NA	0.03	1.00	1.00	NA	NA	NA	NA	NA	0.03
TBMIN	NA	1.00	0.09	0.09	NA	NA	NA	NA	NA	1.00
TBEL50	NA	1.00	1.00	1.00	NA	NA	NA	NA	NA	1.00
SDMAX	NA	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
SDEL50	NA	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

Analysis of variance of the regression indicated statistical significance (Table IX).

Table IX. Analysis of Variance for Multiple Regression of Chlorophyll Response Factor on Various Hydraulic Parameters in Lake Tenkiller.

Parameter	df	SS	MS	F	Significance
Regression	4	40.2	10.0	17.7	$P < 0.01$
Residual	27	15.3	0.568		
Total	31	55.6			

Prediction of the actual chlorophyll trend requires estimation of the CRF and PF. Therefore, a similar regression analysis was performed with the PF. A higher correlation was obtained from regressing the PF on the CRF alone than on the hydraulic parameters. The resulting equation was:

$$PF = -0.00156 \times CRF - 2.90, \quad r^2 = 0.69 \tag{32}$$

The analysis of variance of this regression also indicated signficance (Table X).

Table X. Analysis of Variance for Regression of Power Factor on the Chlorophyll Response Factor in Lake Tenkiller.

Parameter	df	SS	MS	F	Significance
Regression	1	147	147	68.0	$P < 0.01$
Residual	30	64.7	2.16		
Total	31	211			

These regressions were performed on pooled data from the CLP 1992-93 and USACE 1985-86 studies. If the nutrient loads change, say from management, a new regression must be computed. In other words, although the hydraulics are being used to calculate a distribution of chlorophyll model parameters, hydraulics do not cause eutrophication. Only the current nutrient loads apply to these regressions.

<u>Eutrophication Risk Assessment</u>

Assuming hydraulics effected light availability, nutrient fate and transport, and respective biotic linkages as per equation 31, I performed a standard frequency analysis based upon daily values of predicted CRFs (period of record for hydraulics = 1 Jan 81 - 31 Oct 93) from the right side of equation 31. Such a series is often called a full series and should not be confused with the annual series used in most flood-frequency anaylsis (Hewlett 1982). From the resulting probability distribution, I predicted the distribution of the CRF (equation 31) and PF (equation 32). This probability distribution is the intrinsic risk due to morphometric modifiers; the realized risk is the current total P load and is addressed further with TMDLs.

The resulting frequency analysis indicated that the maxima model could describe the chlorophyll *a* trend about 70% of the time, albeit the accuracy declines at ca. 15% exceedance probability (Figure 21). Otherwise stated, the model predicted the lake was in a state of transition such that the hydraulic conditions were not favorable to a maxima trend in chlorophyll *a* for about 30% of the time (Figure 21). I speculate these are conditons of such high flows and low capacities that residence times of N and P fractions are not sufficient for maximum biotic impact, i.e., high chlorophyll densities.

I formulated the model such that the response domain (in Figure 21; x range 0.15 - 0.70) would shift towards the upper right for reservoirs with increasing risks of eutrophy (i.e., higher CRFs with higher probabilities) and towards the lower left for reservoirs with lower risks (i.e., lower CRFs and lower probabilities). Limits of the ordinate were not intended as absolute. Unfortunately, a comparative analysis with other mainstem reservoirs that range in trophic status is yet to be performed and thus I cannot ascertain the relative risk of Lake Tenkiller.

Uncertainty

Every modelling exercise has uncertainty. This one is no exception. Uncertainty that can be quantified describes the risk while that which cannot represents limits to the meaning of the model (see Talcott 1992). In the RERF model, two types of uncertainty can be described, model and parameter.

Model uncertainty is that uncertainty that involves model selection. The sigmoid equation can be justified by the pyramid analogy, while the maxima function is purely empirical. The CRFs were used as a "yardstick" of the chlorophyll *a* trend and do not possess a theoretical foundation (i.e., physical meaning). During inital modelling efforts, I attempted to use a model that substracts turbidity from total P and a ratio of total P to

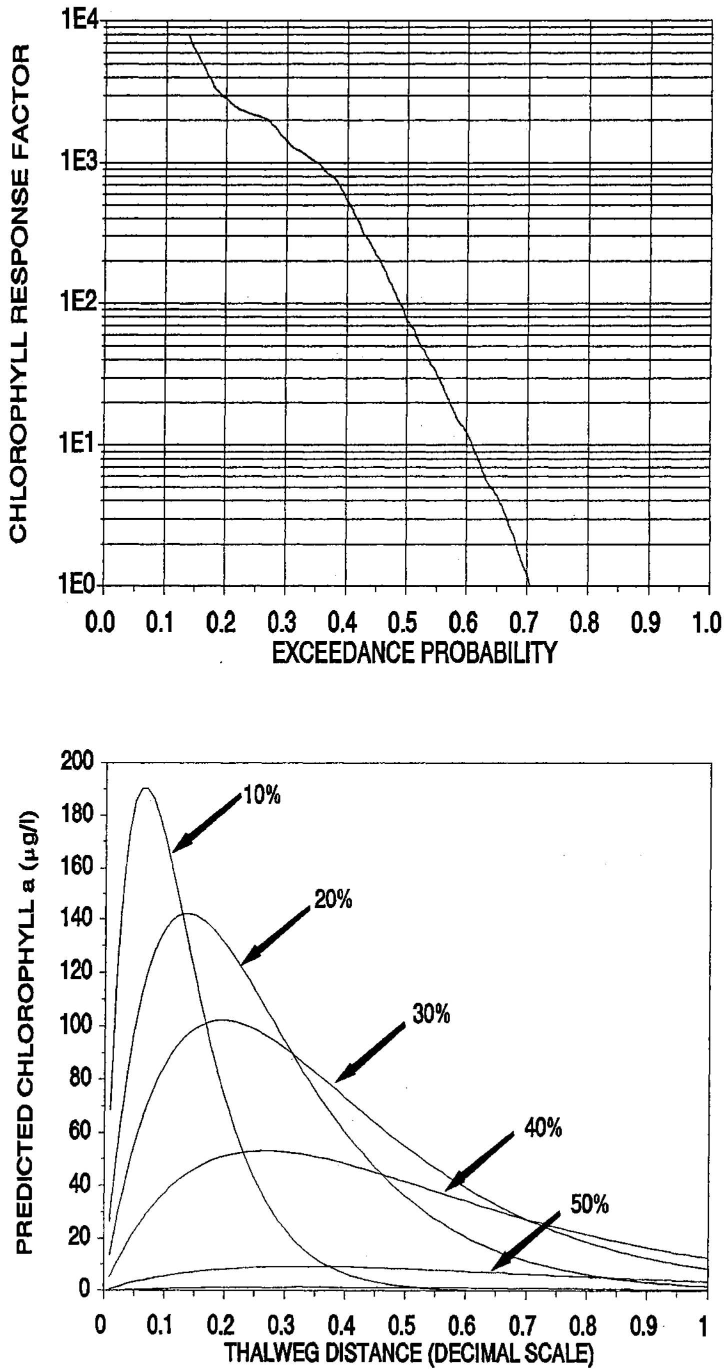

Figure 21. Eutrophication Risk Assesment in Lake Tenkiller from Frequency Analysis of Predicted CRF (top) and Predicted Chlorophyll *a* Trends (bottom).

turbidity. This approach could describe an "optimality" model with real meaning in which the estimated biomass optimizes the parameters. However, initial development using these equations suggested the greatest biomass should occur farther along the thalweg than was observed. The observed maxima occurred at the point of inital decay of turbidity and total P. The observed trend might have been a result of a "threshold" relationship in which the algae were light-limited until the threshold was attained (i.e., initial turbidity decay) and nutrient levels began to impart growth regulation. The maxima, although empirical, yielded a better fit and thus provided a more accurate, though less real, predictor. Model uncertainty also was introduced during application of the multiple regression analysis of hydraulics on natural log transformed CRFs. In this type of uncertainty, quantification of the uncertainty cannot be performed; the relevance is whether the model selected was appropriate.

Parameter uncertainty begins with data collection. Data collection in this study was subjected to a quality assurance/control procedure under which limits of accuracy were set forth. However, the contribution of uncertainty is real. The actual value cannot be evaluated because logistics did not afford multiple replicates of each analysis for every parameter on every sampling event.

Secondly, model parameter uncertainty also was induced from the multiple regressions. This uncertainty can be estimated by the standard errors of the coefficients (Table

XI). These uncertainties were not preserved in the final evaluation.

Table XI. Estimated Parameter Uncertainty for Multiple Regressions in the RERF Model.

Parameter	Coefficient	Standard Error	P-value
Model			
Equation 31			
Constant	-1.09	1.58	0.497
τ_w	0.352	0.188	0.070
Capacity	8.50	1.83	<0.001
Balance	0.206	0.029	<0.001
Inflow	-0.243	0.041	<0.001
Equation 32			
Constant	-2.90	0.278	<0.001
CRF	-1.56E-3	1.90E-4	<0.001

Total Maximum Daily Load

As was discussed previously, I believe reservoirs have intrinsic and extrinsic risk domains. The intrinsic domain is described by the reservoir's response to allochthonous hydraulic conditions. The extrinsic domain is the current stressor load. In eutrophication, the extrinsic domain is the excessive P loads. This extrinsic domain is the manageable variable and requires numeric limitations that promote ecosystem health or prevent further degradation.

Whenever a pollutant limitation is evaluated, as is the case in total maximum daily load (TMDL), an acceptable target value must be put forth. Inherent in this target value is an assumption the ecosystem can tolerate some level of pollutant without undesirable effects which usually include qualitative properties such as aesthetic appearance and/or quantitative properties such as chlorophyll *a* density, species diversity, etc. Odum (1979) called these properties the performance of an ecosystem. I can understand the applicability of this approach to a subsidizing input such as phosphorus. Indeed, aquatic ecosystems require P to exist while too much will cause undesirable effects, i.e., eutrophication. In other words, too little or too much P will cause harm. For a good review on the philosophy of the "stress-subsidy gradient" the reader is referred to Odum (1979). The pertinent question is "What is the maximum level and when does the subsidy become a stressor?" This level becomes the TMDL. However, to answer this question a target ecological endpoint(s) must be defined, managers call these goals or management objectives. This is a difficult task for multipurpose reservoirs.

For example, the recreational user who appreciates water clarity will desire maximum clarity which requires minimizing algal growth in the lacustrine zone. In contrast, fisheries managers desire more algal growth to support a larger warmwater fishery. Hence, conflicts arise. No intention of compromising the various goals is made here. It is my conviction that reversing the accelerated eutrophication of Lake Tenkiller should be a goal.

I calculated a statistical distribution of TMDLs for phosphorus entering Lake Tenkiller using traditional oligotrophy and mesotrophy as target conditions for the TMDL. I compared these "suggested" values with the daily loads observed during the 1974 NES, 1985-86 USACE, and 1992-93 CLP studies.

The model proposed by Vollenweider (1968, 1975, 1976) evaluates permissible and dangerous P loads by inference from a given lake's mean depth divided by the hydraulic retention time (z_a/τ_w, see equation 26). For Lake Tenkiller, the current loads exceeded the critical levels proposed by Vollenweider (1968, 1976) and denoted eutrophy on all sampling events (Table XI).

I compared the current P loading with computed permissible and dangerous P loadings as suggested by Vollenweider (1968, 1975) for each sampling event. Because the hydraulic budget analysis indicated large variations of z_{avg} and τ_w, I included observed hydraulic conditions into the calculations. The estimated TMDL for this method was derived by dividing each statistic of the annual load by 365 (Table XII). I discuss the obvious weakness of this approach later.

Using empirical equations developed by Vighi and Chiaudani (1985) that correlated background (non-anthropogenic) P concentrations with two morphoedaphic indices (conductivity and total alkalinity), I back-calculated an annual load based upon the observed flow and predicted total P concentration derived from equations 27 and 28 for each sampling event during the 1992-93 study (Table XII).

Lake Tenkiller simulations using the model developed by Reckhow (1988) indicated approximately 70% reduction of influent TP would be required to achieve 0.01 μg P/l at the downstream station. The results of these simulations are presented in the project final report. I assumed a 70% reduction in influent total P concentration equated to a 70% reduction in annual load. Based upon this approach, the resulting TMDL for P was somewhat higher than those estimated from methods proposed by Vollenweider (1968, 1975) and Vighi and Chiaudani (1985), although the order of magnitudes were similar (Table XII).

Discussion of the TMDL

A total maximum daily load for phosphorus entering Lake Tenkiller has some inherent weaknesses and should be considered prior to implementing numeric quantities. First, a TMDL assumes that some critical level of phosphorus load is protective of the lake at all times. At times when the lake elevation is low and τ_w is short, the TMDL could be relaxed without significant impacts. In contrast, if the lake elevation is high and a low inflow imparts a long τ_w, the TMDL must be more restrictive for an equally protective effect. Recall, the RERF model predicted these conditions exacerbate eutrophy.

Secondly, seasonal effects influence the impact of

Table XII. Estimated Total Maximum Daily Loads of Total Phosphorus for Lake Tenkiller Headwaters.

	Statistic of TMDL (kg P/d)				
Model	Min	0.25P	0.50P	0.75P	Max
Observed					
EPA-NES 1974	67.5	75.3	113	199	352
USACE 1985-86	109	255	478	645	918
CLP 1992-1993	61.0	150	357	1070	4290
Suggested					
Vollenweider (1968, 1976)					
Oligotrophy	19.7	35.6	70.4	115	210
Mesotrophy	39.3	71.2	141	230	419
Vighi and Chiadauni (1985)					
Conductivity	8.33	13.8	29.6	59.7	212
Alkalinity	10.6	20.7	42.0	92.1	262
Reckhow (1988)					
*Oligotrophy	18.3	45.0	107	321	1287

* = Oligotrophy as TP = 10 μg P/l at station 7.

impinging P loads. During the fall overturn, accumulated hypolimnetic P is transported to the photic zone and a peak in algal growth is usually observed. The implication of this phenomena is dichotomous. First, the fall peak is a period of transition away from P limitation and thus restrictive TMDLs for P is over-protective. In contrast, a more restrictive TMDL could be promoted because this is a period when excess P is inducing ecological release and having its greatest impact (i.e., more P will likely exacerbate eutrophication because a larger biomass is available for uptake).

Thirdly, a TMDL for a specific contaminant does not address factor interactions. Many limnologists agree that the impact of P is highly dependent upon many environmental factors (e.g., available light, micronutrients) and the most important factor for evaluating a P TMDL is concurrent N concentrations. As was discussed earlier, ratios of TN:TP have ecological significance and thus impart a significant interaction with P, i.e., a P TMDL is useful under conditions of TN:TP > ca. 15-20. A lower ratio will magnify the effect of P by promoting blue-green dominance (Shapiro 1973). Should the ratio decrease below critical thresholds ($\approx$ 15), more restrictive TMDLs for P will be required for equal protection. In reality, the ratio is being monitored while TP is being managed. One can argue that the inverse of this scenario is higher nitrogen imparts a protective effect. While low nitrogen levels can foster undesirable blue-green dominance, excessive nitrogen imparts P control of algal growth. Under these conditions, biomass is proportional to P thus implying P control in biomass reduction.

Finally, I cautiously put forth initial trials of TMDL management on a subsidizing pollutant such as P. Almost any ecologist will agree that ecosystems need a baseline input of P for existence. Thus, a TMDL approach seems appropriate. However, when managment of TMDLs is applied to exclusive stressors (e.g., toxic organic that bioacculmulates), I question the applicability of the approach. We may have studies that indicate an ecosystem has innate capacity to accommodate such a toxic insult, but have chronic effects on the evolution of the ecosystem been assessed adequately? The adage "time will tell" seems appropriate.

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APPENDIX A

CLP STUDY 1992-93 LAKE TENKILLER WATER QUALITY DATA

USED FOR RERF DEVELOPMENT

DATE	STATION	Total N mg N/l	Total P mg P/l	TN:TP	Chlr a µg/l	Turb NTU	Secchi m
25-Apr-92	2	ND	0.070	ND	25.3	4.3	0.90
	3	ND	0.052	ND	23.1	4.2	0.88
	4	ND	0.074	ND	19.3	4.8	0.88
	5	ND	0.022	ND	17.1	2.3	1.40
	6	ND	0.013	ND	7.2	1.5	2.50
	7	ND	0.007	ND	3.5	0.6	5.50
04-Jun-92	1	2.38	0.119	20.0	2.8	ND	ND
	2	1.99	0.061	32.8	28.4	ND	0.95
	3	1.86	0.066	28.1	22.3	ND	1.00
	4	2.46	0.061	40.4	34.3	ND	1.00
	5	2.09	0.027	77.3	15.7	ND	1.60
	6	2.71	0.023	116.0	16.5	ND	1.70
	7	3.15	0.016	199.0	9.4	ND	2.40
02-Jul-92	1	2.50	0.111	22.4	10.0	ND	ND
	2	2.28	0.069	33.0	45.2	4.9	0.90
	3	2.13	0.064	33.0	47.7	5.1	0.85
	4	2.08	0.058	35.9	47.2	4.9	0.70
	5	2.00	0.056	35.7	45.6	4.0	1.30
	6	1.73	0.051	33.8	39.6	1.1	1.60
	7	1.99	0.031	65.0	28.0	2.2	1.60
01-Aug-92	1	2.52	0.343	7.3	4.2	18.0	ND
	2	1.09	0.100	10.9	44.5	2.8	0.70
	3	0.92	0.018	50.9	46.2	2.4	0.80
	4	0.97	0.069	14.0	31.0	2.7	0.70
	5	0.51	0.034	15.1	28.0	1.6	1.23
	6	0.70	0.020	34.9	13.3	1.2	1.65
	7	0.63	0.014	45.3	12.1	1.2	1.70
19-Aug-92	1	1.21	0.080	15.1	3.9	6.6	ND
	2	0.77	0.063	12.3	32.7	5.9	1.28
	3	0.76	0.081	9.4	32.9	5.1	0.90
	4	0.81	0.069	11.8	33.4	5.2	0.90
	5	0.57	0.057	9.9	18.8	2.6	1.50
	6	0.31	0.042	7.5	13.4	2.3	1.70
	7	0.39	0.038	10.3	10.1	1.5	2.20
12-Sep-92	1	1.33	0.097	13.7	1.1	8.7	1.00
	2	0.99	0.084	11.8	43.4	6.6	0.80
	3	0.72	0.081	8.8	33.0	13.5	0.80
	4	0.59	0.057	10.4	29.2	10.1	1.00
	5	0.37	0.041	9.0	15.8	5.4	1.60
	6	0.47	0.017	27.5	11.0	4.4	1.80
	7	0.31	0.025	12.5	6.3	3.0	2.30
24-Oct-92	1	1.58	0.086	18.4	1.9	ND	ND
	2	1.16	0.048	24.2	40.3	3.5	1.30
	3	0.97	0.031	31.4	26.1	2.0	1.90
	4	1.17	0.039	30.0	23.4	5.3	1.90
	5	0.75	0.025	30.0	11.8	2.0	2.30
	6	0.76	0.023	32.8	14.2	1.7	2.60
	7	0.61	0.023	26.6	12.6	1.3	4.30
08-Mar-93	1	2.86	0.085	33.7	1.0	ND	ND
	2	1.47	0.073	20.1	2.7	ND	0.70
	3	1.69	0.079	21.4	3.9	ND	0.75
	4	1.86	0.081	23.0	4.3	ND	0.80
	5	1.94	0.118	16.5	4.1	ND	0.30
	6	1.80	0.085	21.2	9.4	ND	1.70
	7	1.42	0.087	16.3	2.5	ND	1.45
18-Apr-93	1	2.36	0.287	8.2	2.6	ND	ND
	2	1.71	0.159	10.8	1.6	ND	0.30
	3	1.50	0.146	10.3	8.9	ND	0.27
	4	2.00	0.176	11.4	2.7	ND	0.30
	5	1.83	0.124	14.8	5.8	ND	0.48
	6	1.66	0.078	21.3	11.9	ND	1.20
	7	1.93	0.067	28.8	30.6	ND	1.40
26-May-93	1	2.52	0.106	23.8	71.1	17.0	ND

(cont.)

DATE	STATION	Total N mg N/l	Total P mg P/l	TN:TP	Chlr a µg/l	Turb NTU	Secchi m
26-May-93	2	1.36	0.048	28.3	44.4	13.0	1.10
	3	1.24	0.054	23.0	29.4	15.0	1.30
	4	1.51	0.056	26.9	36.4	7.4	1.20
	5	1.39	0.056	25.0	15.5	8.5	1.40
	6	1.36	0.046	29.5	5.8	8.0	2.00
	7	1.37	0.037	37.0	8.5	4.8	2.20
25-Jun-93	1	1.92	0.131	14.7	1.7	39.0	ND
	2	1.08	0.102	10.6	46.8	28.0	1.20
	3	1.01	0.097	10.4	45.0	56.0	1.10
	4	0.90	0.081	11.1	38.0	ND	0.95
	5	0.79	0.064	12.3	40.3	28.0	1.15
	6	0.91	0.040	22.8	32.4	23.0	1.55
	7	0.87	0.033	26.4	39.1	24.0	1.40
22-Jul-93	1	1.67	0.130	12.8	2.5	6.5	ND
	2	0.81	0.046	17.6	10.6	6.3	1.20
	3	0.80	0.051	15.7	12.0	8.3	1.10
	4	0.72	0.059	12.2	21.7	6.3	1.20
	5	0.49	0.038	12.9	12.6	4.9	1.50
	6	0.74	0.022	33.6	3.7	2.9	2.10
	7	0.74	0.020	37.0	6.4	4.7	2.00
04-Aug-93	2	0.97	0.076	12.7	20.0	14.0	0.70
	3	1.23	0.076	16.1	17.3	18.0	0.90
	4	1.34	0.076	17.6	22.9	15.0	0.85
	5	0.61	0.042	14.4	26.5	10.0	1.20
	6	0.64	0.023	28.1	13.8	4.3	2.10
	7	0.77	0.016	48.3	8.2	4.4	3.00
19-Aug-93	1	1.57	0.129	12.2	21.0	2.7	ND
	2	0.83	0.090	9.2	35.4	4.9	1.00
	3	0.61	0.061	10.0	26.6	4.9	1.10
	4	0.47	0.050	9.4	31.1	3.9	1.30
	5	0.50	0.034	14.7	16.9	2.2	1.95
	6	0.62	0.028	22.3	10.5	2.9	2.50
	7	0.70	0.021	33.3	7.0	1.5	2.80
02-Sep-93	1	0.92	0.125	7.4	1.2	7.0	ND
	2	0.71	0.083	8.6	25.7	7.4	0.80
	3	0.77	0.093	8.3	39.6	7.5	0.85
	4	0.56	0.086	6.5	28.1	7.2	0.80
	5	0.45	0.041	11.0	18.9	2.6	1.40
	6	0.35	0.015	23.1	22.3	2.2	2.30
	7	0.41	0.015	27.3	24.9	1.6	2.00
16-Sep-93	1	5.22	0.178	29.3	2.0	25.0	ND
	2	3.82	0.119	32.1	8.6	45.2	0.20
	3	2.17	0.103	21.1	30.9	17.0	0.50
	4	2.58	0.165	15.6	26.0	23.5	0.45
	5	1.55	0.067	23.1	23.5	9.5	1.00
	6	0.78	0.013	60.0	11.6	1.8	2.40
	7	1.03	0.009	114.4	10.7	2.4	3.10
30-Sep-93	1	3.40	0.096	35.4	0.8	4.4	ND
	2	2.00	0.075	26.7	35.2	6.1	0.80
	3	3.78	0.223	17.0	44.5	41.0	0.80
	4	1.80	0.075	24.0	33.9	11.0	0.90
	5	1.33	0.059	22.5	13.7	4.5	1.40
	6	0.58	0.164	3.5	8.9	2.2	2.80
	7	0.67	0.051	13.1	5.7	2.1	3.30
21-Oct-93	1	2.00	0.157	12.7	2.9	15.5	ND
	2	1.59	0.101	15.7	3.8	13.8	0.60
	3	1.61	0.090	17.9	5.6	13.9	0.70
	4	1.03	0.056	18.4	9.5	5.3	1.10
	5	0.85	0.051	16.7	11.4	5.1	2.20
	6	0.61	0.023	26.5	5.3	3.2	2.10
	7	0.61	0.017	35.9	1.3	1.8	2.80

APPENDIX B

RERF MODEL PARAMETERS BY DATE

FOR LAKE TENKILLER CLP STUDY 1992-93

DATE	MODEL	MAX[1]	MIN[1]	SF	EL50[2]	RED[3]	CRF	PF	PEAK DIST[2]
25 Apr 92	TURB	4.8	0.76	-10.1	0.50	84	--	--	--
	TP	70.0	8.37	-13.7	0.44	88	--	--	--
	SD	66.7	9.00	8.53	0.83	86	--	--	--
	CHLR	--	--	--	--	--	290	-4.46	0.22
04 Jun 92	TURB	NC	NC	NC	NC	100	--	--	--
	TP	119	19.6	-9.79	0.26	84	--	--	--
	SD	26.2	9.50	6.19	0.70	64	--	--	--
	CHLR	--	--	--	--	--	266	-3.57	0.28
02 Jul 92	TURB	NC	NC	NC	NC	NC	--	--	--
	TP	111	45.5	-14.1	0.22	59	--	--	--
	SD	16.3	7.00	11.8	0.48	57	--	--	--
	CHLR	--	--	--	--	--	339	-2.54	0.39
01 Aug 92	TURB	18.0	NC	NC	NC	100	--	--	--
	TP	343	0.33	-3.88	NC	100	--	--	--
	SD	16.9	7.00	16.6	0.52	59	--	--	--
	CHLR	--	--	--	--	--	511	-4.42	0.23
19 Aug 92	TURB	6.6	1.73	-10.3	0.40	74	--	--	--
	TP	80.0	35.3	-6.71	0.49	56	--	--	--
	SD	23.7	9.00	5.84	0.66	62	--	--	--
	CHLR	--	--	--	--	--	375	4.16	0.24
12 Sep 92	TURB	13.5	0.00	-2.86	0.52	100	--	--	--
	TP	97.0	22.0	-10.6	0.37	77	--	--	--
	SD	22.8	8.00	8.04	0.56	65	--	--	--
	CHLR	--	--	--	--	--	559	-5.28	0.19
24 Oct 92	TURB	5.3	0.79	-3.79	0.39	85	--	--	--
	TP	86.0	24.2	-15.4	0.21	72	--	--	--
	SD	73.4	13.0	3.80	1.00	82	--	--	--
	CHLR	--	--	--	--	--	373	-4.48	0.22
08 Mar 93	TURB	NC	NC	NC	NC	100	--	--	--
	TP	118	0.13	2.16	NC	100	--	--	--
	SD	15.8	7.00	224	0.62	56	--	--	--
	CHLR	--	--	--	--	--	22	-1.51	0.66
18 Apr 93	TURB	NC	NC	NC	NC	100	--	--	--
	TP	287	78.6	-7.83	0.25	73	--	--	--
	SD	14.1	2.70	15.50	0.62	81	--	--	--

(cont.)

DATE	MODEL	MAX	MIN	SF	EL50	RED[3]	CRF	PF	PEAK DIST
18 Apr 93	CHLR	--	--	--	--	--	5	1.80	1.00
26 May 93	TURB	17.0	6.97	-46.1	0.29	59	--	--	--
	TP	106	48.3	-25.5	0.16	54	--	--	--
	SD	22.5	11.0	9.66	0.60	51	--	--	--
	CHLR	--	--	--	--	--	8068	-14.7	0.01
25 Jun 93	TURB	56.0	NC	NC	NC	100	--	--	--
	TP	131	33.1	-7.03	0.37	75	--	--	--
	SD	14.8	9.50	127.4	0.53	36	--	--	--
	CHLR	--	--	--	--	--	274	-2.23	0.45
22 Jul 93	TURB	8.3	3.76	-8.15	0.38	55	--	--	--
	TP	130	28.1	-13.3	0.19	78	--	--	--
	SD	20.6	11.0	19.58	0.54	47	--	--	--
	CHLR	--	--	--	--	--	138	-3.49	0.29
04 Aug 93	TURB	18.0	3.71	-9.17	0.48	79	--	--	--
	TP	76.0	18.9	-21.5	0.51	75	--	--	--
	SD	32.2	7.00	8.02	0.69	78	--	--	--
	CHLR	--	--	--	--	--	164	-2.77	0.36
19 Aug 93	TURB	4.9	0.00	-2.86	0.75	100	--	--	--
	TP	129	27.7	-21.4	0.26	79	--	--	--
	SD	27.7	10.0	10.2	0.52	64	--	--	--
	CHLR	--	--	--	--	--	538	-5.44	0.18
02 Sep 93	TURB	7.5	1.91	-25.2	0.45	75	--	--	--
	TP	125	11.4	-7.86	0.38	91	--	--	--
	SD	21.5	8.00	141	0.53	63	--	--	--
	CHLR	--	--	--	--	--	208	2.53	0.40
16 Sep 93	TURB	45.2	0.00	-4.54	0.32	100	--	--	--
	TP	165	0.00	-6.68	0.44	100	--	--	--
	SD	32.1	2.00	9.49	0.62	94	--	--	--
	CHLR	--	--	--	--	--	177	-2.90	0.34
30 Sep 93	TURB	41.0	NC	-2.33	NC	100	--	--	--
	TP	96.0	0.00	-0.68	0.30	100	--	--	--
	SD	33.2	8.00	13.1	0.61	76	--	--	--
	CHLR	--	--	--	--	--	650	-5.62	0.18
21 Oct 93	TURB	15.5	3.35	-55.3	0.30	78	--	--	--

(cont.)

DATE	MODEL	MAX[1]	MIN[1]	SF	EL50[2]	RED[3]	CRF	PF	PEAK DIST[2]
21 Oct 93	TP	157	29.5	-19.7	0.27	81	--	--	--
	SD	24.6	6.00	16.7	0.41	76	--	--	--
	CHLR	--	--	--	--	--	54.0	-2.65	0.38

NC = non-convergence.
[1] = P in µg P/l, TURB in NTU, Secchi Disk in dm, and chlorophyll *a* in µg/l.
[2] = EL50 and peak distance are expressed as decimal fraction of thalweg distance.
[3] = RED is % reduction in parameter (i.e., 100*(MAX-MIN)/MAX).

VITA

Noble Jobe, III

Candidate for the Degree of

Doctor of Philosophy

Thesis: RISK BASED TROPHIC STATUS ANALYSIS IN RESERVOIRS

Major Field: Zoology

Biographical:

Personal Data: Born in Midwest City, Oklahoma, December, 29, 1963, the son of Noble Jobe, Jr. and JoAnn Bacon.

Education: Graduated Choctaw High School, Choctaw, Oklahoma, May 1982, received Associate of Arts degree in Secondary Education from Rose State College, Midwest City, Oklahoma, May 1987; received Bachelor of Science degree in Secondary Education from East Central University, Ada, Oklahoma, May 1989; received Master of Science degree at Oklahoma State University, Stillwater, Oklahoma, December 1991; completed requirements for the Doctor of Philosophy degree at Oklahoma State University in May 1995.

Professional Experience: Research Assistant, Water Quality Research Lab, Oklahoma State University, June, 1989 to August, 1989; Teaching Assistant, Department of Zoology, Oklahoma State University, August, 1989 to May, 1990; Research Assistant, Water Quality Research Lab, Oklahoma State University, June, 1990 to August, 1990; Teaching Assistant, Department of Zoology, Oklahoma State University, August, 1990 to May, 1991; Research Assistant, Water Quality Research Lab, Oklahoma State University, June, 1991 to August, 1991; Teaching Assistant, Department of Zoology, Oklahoma State University, August, 1991 to December, 1991; Research Associate, Water Quality Research Lab, Oklahoma State University, January, 1992 to present.