EUTROPHICATION MONITORING AND PREDICTION

DISSERTATION

Presented to the Graduate Council of the
University of North Texas in Partial
Fulfillment of the Requirements

For the Degree of

DOCTOR OF PHILOSOPHY

By

Stefan Hugh Cairns, B.A., B.S., M.S.

Denton, Texas

December, 1993

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Changes in trophic status are often related to increases or decreases in the allocthonous inputs of nutrients from changes in land use and management practices. Lake and reservoir managers are continually faced with the questions of what to monitor, how to monitor it, and how much change is necessary to be considered significant. This study is a compilation of four manuscripts, addressing one of these questions, using data from six reservoirs in Texas. Chapter II evaluates trophic status endpoints agreement and selection. Chapter III addresses the validation of eutrophication model predictions. Chapter IV evaluates the efficacy of using satellite reflectance data for monitoring trophic status. Chapter V evaluates the development of a chlorophyll a variability model.

Chlorophyll a concentrations and Secchi disk depth appear to be the most effective trophic status indicators for managers of lakes and reservoirs to monitor because of the agreement in their description of trophic status. Chlorophyll a concentrations would be a better direct indicator of the turbidity associated with phytoplankton

biomass.

The mean summer epilimnetic chlorophyll _a_ eutrophication model predictions of Ray Roberts Lake were higher than measured concentrations two years after the impoundment of Ray Roberts Lake. The predicted high and low range values, determined by the high and low nutrient values from the nutrient loading model, were also higher than the measured chlorophyll _a_ concentrations.

Low correlation between water clarity parameter values and reflectance values appeared to relate directly to the magnitude of difference in chlorophyll _a_ and turbidity values. The use of remotely sensed data appears not to be practical for lake monitoring in those cases where detectable differences are less than the magnitude of difference needed for statistical significance.

The development of a preliminary chlorophyll _a_ variability model has shown potential for standardizing lake managers judgement of statistical significant difference in changes in chlorophyll _a_ concentrations. The benefit in using this model approach for this purpose is that many monitoring programs depend on minimal number of samples to determine significant changes.

PREFACE

This dissertation is a compilation of four papers submitted for publication. Chapter II through Chapter V are the papers submitted and are complete in themselves.

A portion of the work presented in this dissertation was supported by a research grant from the United States Army Corps of Engineers.

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CHAPTER I

INTRODUCTION

Water is a requirement of all organisms in the biosphere. The total volume of water in the biosphere has been estimated at $1,380,000 \times 10^{12}$ cubic meters in the oceans, 230×10^{12} in the surface waters, $26,000 \times 10^{12}$ in glaciers, and $7,000 \times 10^{12}$ in groundwater. The quality of the water required by organisms varies with each organism and the type of water in which that species has evolved. As organisms vary considerably in their tolerances, specialized faunas and floras have evolved with the differing water quality conditions (Patrick 1986). Man has historically used surface waters as the main source for drinking, agriculture, industry, and recreation, as well as, the main recipients of the industrial and municipal effluents. These activities have their own water quality requirements and effects on the receiving waters.

Increased population with its associated industrial growth, intensification of agricultural production, river-basin development, and recreational use of waters, have made the problem of pollution as complex as it is critical. Human activities, which introduce excess nutrients, along with other pollutants, into lakes, streams, and estuaries, have caused significant changes in aquatic environments. Excess nutrients greatly accelerate the process of eutrophication. Accelerated eutrophication changes the flora and fauna of the system. These changes often interfere with the uses and asthethics of the water resource. Excessive nutrient loadings in reservoirs and lakes result in limits on recreational use and increases in water treatment cost.

A better understanding of the eutrophication process will improve the ability to manage water as a multi-use resource. An understanding of the variability associated with eutrophication endpoints, such as chlorophyll _a_ and light penetration, will improve the ability to monitor water bodies and detect significant changes due to management practices. Successful lake restoration projects such as Medical Lake, Washington (Soltero _et al._, 1981; Scholz _et al._, 1985), Lake Washington (Edmunson, 1991), and Shagawa Lake, Minnesota (Larsen _et al._, 1981) have demonstrated the benefits gained from understanding the eutrophication process. With the development of the ecoregion concept there is a continuing need to determine effective endpoints to measure or predict eutrophication within each region. Additionally, such understanding will aid in the fulfillment of section 314a of the Federal Water Pollution Control Act of 1972 (Public Law 92-500) which requires the classification of water bodies by each state according to the level of eutrophication. The National Research Council (1992) concluded that one area of research needed for lake restoration was:

> "Improved assessment programs are needed to determine the severity and extent of damage in lakes and their change in status over time. Innovative basic research is required to improve the science of assessment and monitoring. There is a great need for cost-effective, reliable indicators of ecosystem function, including those that will reflect long-term change and response to stress. Research on indicators should include traditional community and ecosystem measurements, paleoecological trend assessments, and remote sensing."

Historical Overview

The terms "eutrophic," "mesotrophic," and "oligotrophic" were introduced by Weber in 1907 to describe nutrient conditions in German bogs. Plants characteristic of eutrophic bogs were termed eutraphent, or well-nourished. Weber used the terms to describe the succession of the bogs from eutrophic condition to oligotrophic condition due to the build up of continually leached soils (Hutchinson, 1969). Naumann (1919) introduced Weber's terms into limnology to describe a theoretical classification scheme of water types. The practical criterion of Naumann was that when a lake contained eutrophic water the phytoplankton made the water turbid for most of the year while lakes containing oligotrophic water did not (Hutchinson, 1969). Tieling (1916) characterized the different phytoplankton communities associated with the different water types and related this to various geographical regions. Thienemann (1925) correlated hypolimnetic oxygen concentrations, nutrient concentrations, and depth to both the benthic fauna, as well as, phytoplankton blooms. In 1931, Juday and Birge determined that phytoplankton populations were sometimes maintained, or even increased, with loss of phosphorus in the water. Pearsall (1932) pointed out that bluegreen blooms, characteristic of eutrophic lakes, actually appeared when there was a noticeable nutrient deficiency in the water. Sawyer (1947) demonstrated the relationship of spring nutrient concentrations and summer algal blooms. Hutchinson (1941) identified the cyclic nature of nutrients in sediments, phytoplankton, and the water column (Hutchinson, 1969).

During the 1960's and 1970's, many large scale research programs were funded, mostly by federal agencies, on eutrophication. These research programs identified causes of eutrophication, relationships of nutrient loading to water column responses, and restoration methodology.

Classic works such as Edmunson (1969), Hutchinson (1975), Shapiro (1973), Wetzel (1966) are but a few of the researchers who addressed the problem of eutrophication. Beginning in the 1970's and continuing into the present, trophic status evaluation has been aided by the development of trophic indices and mathematical models. The most widely used trophic indices are those developed by Carlson (1977) which are derived from values of Secchi disk transparency, total phosphorus concentration, and chlorophyll a concentration. The strong correlations of each of these parameters has been well documented and each parameter is individually a good reflection of lake and reservoir physical, chemical, and biological trophic status. The indices presented by Carlson were one of the first attempts to standardize the eutrophication endpoints by converting each parameter into standard trophic state indicator units. Recently more complex mathematical models have been developed to describe and predict trophic status based on multiparameter relationships, both internal and external, of a lake system. An extensive description of many of the commonly used lake and reservoir models is provided by Svatos (1986) and Armstrong, et al., (1987).

Vollenweider (1976) used total phosphorus in conjunction with mean depth and residence time to predict mean epilimnetic chlorophyll a concentration. This model provides the basis for the Organization for Economic

Cooperation and Development (OECD) eutrophication model described by Rast and Lee (1983) and Vollenweider and Kerekes (1981). Pillard (1988) used nutrient loading relationships as the best predictor of trophic status in his pre-impoundment study of Ray Roberts Lake. Pillard (1988) used the OECD eutrophication model in conjunction with land use analysis from satellite reflectance data to predict nutrient loading and chlorophyll a response in Lake Ray Roberts.

The use of remote sensing systems to predict and monitor both land use and water quality has greatly increased over the past 15 years. Remote sensing systems measure reflectance as it varies with spectral wavelength. The result is an index of color which can the be used for the quantification of land use or any water parameter that influences light reflectance or backscattering. Richie et al. (1987) determined that Landsat Multispectral Scanner (MSS) data effectively estimate suspended sediments in aquatic systems where the mean annual concentrations were greater than 5.0 mg/L. Lathrop and Lillesand (1986) found significant relationships between Thematic Mapper (TM) data and Secchi disk depth, chlorophyll a concentrations, turbidity, and surface temperature in Green Bay and Central Lake Michigan. In 1989 Lathrop and Lillesand also found significant correlations of Systeme Pour l'Observation de la Terra (SPOT) multispectral data with the same water quality parameters, with the exception of temperature which can not be measured by the SPOT-1 system. Both Lillesand et al. (1983), using Landsat data, and Wezernak et al. (1976), using low altitude M-7 multispectral scanner, determined that remotely sensed data could effectively predict trophic status of inland water bodies.

Models are used to develop a better understanding of a system and its response to external influences, and to aid resource managers in monitoring and restoration decisions. Butkus (1990) states that some measure of uncertainty must be defined for results to be useful for decision making. Model predictions are typically deterministic yielding a single value for a specified set of conditions. Prediction ranges have been commonly determined by application of "best case - worst case" values to the model.

Study Overview

The objective of this research was to determine the efficacy of several methods for the prediction and monitoring of the process of eutrophication. Selected trophic status endpoints of impoundments were compared to determine the level of agreement and to determine their statistical, ecological, and social relevance. The prediction sensitivity of a selected mathematical eutrophication model was determined by examining the level of agreement between the predicted mean and range for mean summer epilimnetic chlorophyll a concentrations with measured values in the same impoundment two years after predictions were made. Efficacy of using remotely sensed satellite reflectance data to predict and monitor water quality was determined by the level of agreement with measured water quality parameters of three impoundments in the same drainage area. Finally, chlorophyll a data from six impoundments were used to develop a chlorophyll a variance model. The model was based on the Minimum Detectable Difference (MDD) calculated and combined at different temporal scales to determine the influence of time and sample size on the agreement between variability and chlorophyll a concentrations. The MDD is a ratio of the variance and sample size at set or calculated probability levels. The 95% confidence limits about the relationship were used as the "Chlorophyll Minimum Detection Window" (CMDW). The efficacy of the CMDW as a monitoring tool was determined by comparison to the results of a eutrophication prediction model (Organization for Economic Cooperation and Development), used in the interpretation of remotely sensed data, and its degree of similarity to successful lake restoration cases. Thus the primary objective of this research was to determine the efficacy of standardizing the eutrophication endpoint (chlorophyll a) and the mathmatical significance of changes in chlorophyll a.

Hypotheses

It is apparent from the literature that the description and prediction of eutrophication endpoints can be both effectively and efficiently accomplished with the use of a wide array of tools, indices, mathematical models, and remotely sensed data. Considering the primary objectives of this study, the following hypotheses were formulated and tested.

Ho1 - There is no relationship between the Minimum Dectectable Difference (MDD) of chlorophyll a data and its associated mean concentration values.

Ho2 - There are no differences in the predicted mean summer epilimnetic chlorophyll a values for Ray Roberts Lake pre-impoundment prediction and measured chlorophyll a values from Ray Roberts Lake two years after impoundment. Trophic status predictions were based on values derived from the OECD (Organization for Economic Cooperation and Development) eutrophication model.

Ho3 - There is no relationship between remotely sensed reflectance values and trophic status indicators.

Chapters II, III, IV, and V each address a part of the primary objectives. These chapters each represent a manuscript dealing with one aspect of monitoring the process of eutrophication. Chapter II addresses the question of which eutrophication endpoints to consider for monitoring the eutrophication proccess. Chapter III addresses the predictive capability of the empirical mathmatical eutrophication model described in Ho2. Chapter IV addresses the eficacy of using satelite imagery for monitoring the eutrophication process (Ho3). Chapter V addresses the development of a chlorophyll a variability model (Ho1) as a management aid in the determination of statistical significant difference.

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CHAPTER II

ASSESSMENT OF TROPHIC STATUS INDICATORS

Introduction

Changes related to excessive nutrient loading are a serious problem in many reservoirs which can result in limits on recreational use and increases in treatment cost if used as a drinking water source. An understanding of the relationship of eutrophication endpoints will improve the ability to monitor water bodies and detect significant changes due to management practices. Successful lake restoration projects such as Medical Lake, Washington (Soltero *et al.*, 1981; Scholz *et al.*, 1985), Lake Washington (Edmunson, 1991), and Shagawa Lake, Minnesota (Larsen *et al.*, 1981) have emphasized the benefits gained from the understanding of the eutrophication process. The National Research Council (1992) concluded that one area of research needed for lake restoration was:

> "Improved assessment programs are needed to determine the severity and extent of damage in lakes and their change in status over time. Innovative basic research is required to improve the science of assessment and monitoring. There is a great need for cost-effective, reliable

indicators of ecosystem function, including those that will reflect long-term change and response to stress. Research on indicators should include traditional community and ecosystem measurements, paleoecological trend assessments, and remote sensing."

The term eutrophic, meaning "well nourished", was first introduced by Weber in 1907 to describe the condition of bogs (Hutchinson, 1969). From that time on eutrophic, mesotrophic, and oligotrophic have been used to describe the levels of potential productivity (nutrients) and observed productivity in surface waters. Classic works such as Edmunson (1969), Hutchinson (1975), Shapiro (1973), Wetzel (1966) are but a few of the researchers addressing the problem of eutrophication. Beginning in the 1970's and continuing into the present, trophic status evaluation has been aided by the development of trophic indices and mathematical models. The most widely used trophic indices are those developed by Carlson (1977) which are derived from values of Secchi disk transparency, total phosphorus concentration, and chlorophyll a concentration. The strong correlation of each of these parameters has been well documented and each parameter is individually a good reflection of their respective physical, chemical, and biological trophic status. These indices, however, differ in the levels at which the eutrophication labels are placed. The broadness of the labels and the differences of level demarcation make trophic description and monitoring difficult. The development of a consistent trophic scale by Carlson (1977) provided a means by which many trophic indices may be compared without the use of an eutrophication classification. Other trophic status indices, such as Nygaard's phytoplankton composition (EPA,1977), can not be directly converted to the standardized units developed by Carlson. These indices must be compared in the context of the broader trophic classifications.

Recently more complex mathematical models have been developed to describe and predict trophic status based on multiparameter relationships, both internal and external, of the lake system itself. An extensive description of many of the commonly used lake and reservoir models is provided by Svatos (1986) and Armstrong, et al., (1987). The Organization for Economic Cooperation and Development (OECD) eutrophication model based on Vollenweider's normalized phosphorus loading is recommended in the U.S. EPA Quality Criteria for Water and determined most applicable to Texas lakes by Armstrong, et al., (1987). Lee and Jones (1992) showed a strong relationship between this nutrient loading and effects in the fisheries of the water body. Generally these models are useful as tools for the prediction of eutrophication endpoints with changing management practices.

The successful monitoring of eutrophication in lakes is dependent on the choice of eutrophication endpoint. The endpoint(s) should be both ecologically and socially important and have the ability to be measured with enough sensitivity to detect significant change (NRC, 1992). The significance of the change is most often determined by the relationship within any given data set. Indices do not, however, give any framework with which to judge significance or to determine the appropriateness of the endpoint chosen for a given water body. For eutrophication endpoints to have practical application for monitoring change, the sensitivity of the endpoint's measurements needs to be judged by a consistent measure. A model's predicted mean and range of an endpoint would also need to have a fine enough resolution to reflect significant change at a given trophic state.

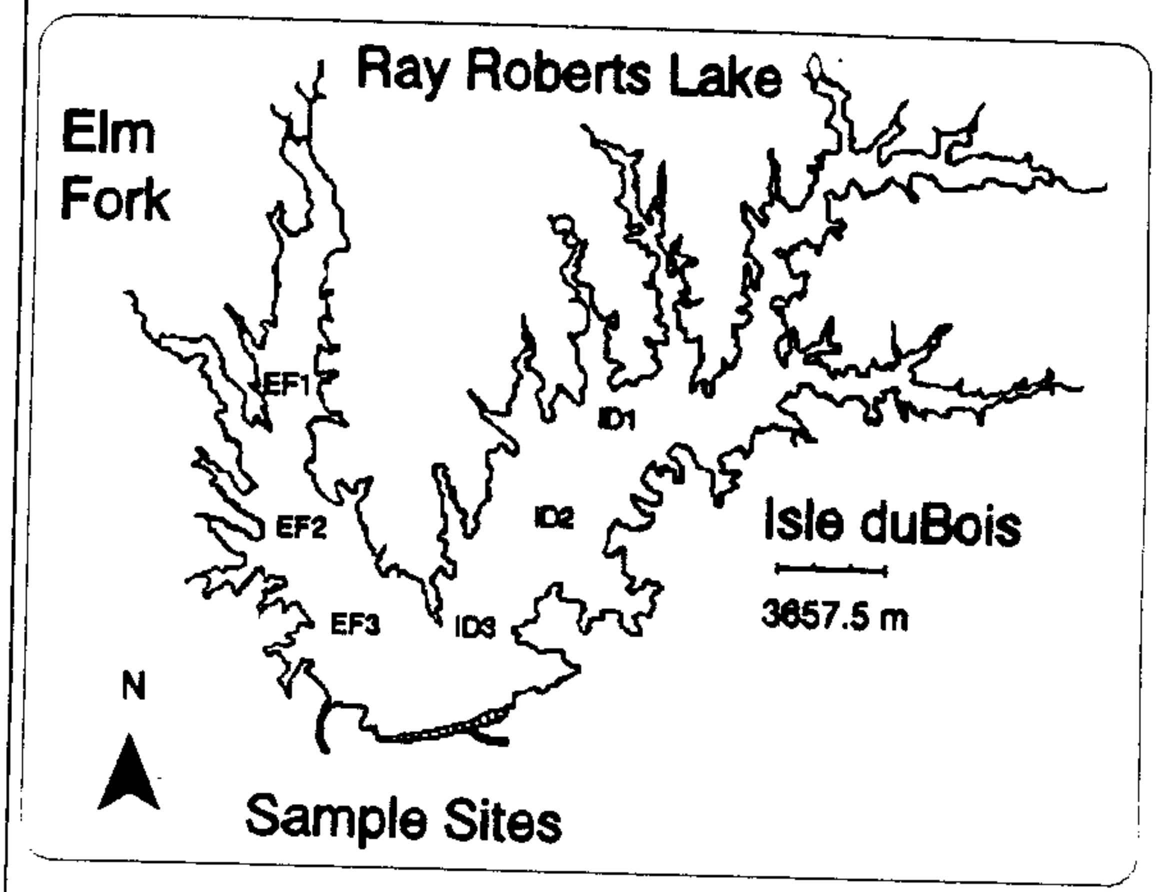
It is the purpose in this paper to compare the level of agreement between trophic status indices for selected water bodies in Texas with emphasis on Ray Roberts Lake, newly impounded in 1987 (IAS, 1992). The trophic status indicators were compared and analyzed to determine the most applicable endpoints for the systems sampled.

Study Sites

Ray Roberts Lake

Construction of Ray Roberts Lake, under the control of the Army Corps of Engineers, was initiated in 1981 and completed in 1987 (USACOE, 1983). Ray Roberts Lake is located in Denton County extends into Cooke County in the Elm Fork Arm of the Trinity River and into Grayson County in the Isle du Bois Arm of the Trinity River. Ray Roberts Dam is an earthen dam located 96.6 km (river mile 60.0) on the Elm Fork of the Trinity River, 48.3 km (30.0 river miles) upstream of the existing Lewisville Dam, and approximately 16.1 km (10 miles) north of the city of Denton. The reservoir covers 11,878 hectares (29,350 acres) at the Conservation Pool and 14,933 hectares (36,900 acres) at the Flood Control Pool (Map 1).

Map 1 -- Location of Ray Roberts Lake sampling sites for monthly water quality parameters and quarterly plankton parameters from June 1989 to May 1990


Ray Roberts Lake
Elm Fork
EF1
EF2
EF3
ID1
ID2
ID3
Isle duBois
3657.5 m
N
Sample Sites

Stockpond 1

Stockpond 1 is a tertiary lagoon system for an industrial plant on the Texas coast. This pond is a hypereutrophic appearing system. The water in the last section of the tertiary lagoon and the settling pond is "pea soup" green from planktonic algae. The littoral zone has productive populations of shrimp and aquatic snails. The shoreline also has large numbers of feeding egrets and herons. The primary objective of the lagoon system is the physical, chemical, and biological treatment of the effluent from the industrial plant.

Stockpond 2

Stockpond 2 is a small earthen dam impoundment located near Dallas, Texas. Stockpond 2 is spring fed and receives runoff from the lawns around a large house located within 100 yards of the pond. Visually the pond appears to be highly eutrophic with dense mats of filamentous algae in the littoral zone and with anoxic odors present.

Stockpond 3

Stockpond 3 is a small earthen dam impoundment located near Stockpond 2 and is separated from Stockpond 2 drainage area by a ridge. Stockpond 3 receives water only from ground water infiltration and surface runoff from a predominantly timbered area. This pond appears much less eutrophic than Stockpond 2 although the littoral zone is dominated by macrophytes.

Sampling

Triplicate water samples were collected monthly from six sites in the limnetic, open water, zones of Lake Ray Roberts (Figure 1) from June 1989 to May 1990. Samples were analyzed for alkalinity, hardness, chloride, ammonia, nitrate, nitrite, total phosphate, orthophosphate, suspended solids, dissolved solids, chlorophyll _a_, pheophytin _a_, turbidity, sulfate, temperature, dissolved oxygen, pH, secchi disk depth, 1% light extinction, and conductivity following methods in American Public Health Association (1985). Phytoplankton community structure was analyzed quarterly during the same time period.

Stockpond 1 was sampled on three successive days (September 28,29 and 30, 1990). Stockponds 2 and 3 were sampled on August 16, 1991. Sampling of the three stockponds were part of congruent studies. Water analysis included all the parameters described for Ray Roberts Lake monthly sampling.

Results and Discussion

Carlson (1977) developed the Trophic State Index (TSI) for values of secchi disk transparency, total phosphorus, and chlorophyll a concentrations. TSI values for each parameter were scaled between 0 and 100. Carlson gave eutrophic status to those values of 50 or more. Chlorophyll a concentrations of greater than 6.4 ug/L described a eutrophic condition, and 10 TSI units approximated a doubling of the concentration. The application of a trophic status label, such as eutrophic, has been attached to different levels by different authors. Since consistency is of primary importance rather than the level itself, the delineation provided by the National Eutrophication Survey (NES) was used as descriptors of these three trophic indicators (Table 1). The measured values of each indicator for each system were compared to each other after calculating the appropriate TSI value using simple regression modeling of Carlson's index. This was done to insure direct comparability of values by standardizing the units.

Table 1.- Trophic state indicator values (ranges) and the National Euutrophication Survey's trophic state label.

Trophic State	Chlorophyll a (ug/L)	Total Phosphorus (ug/L)	Secchi Disk meters
Oligotrophic	< 7	< 10	> 3.7
Mesotrophic	7 - 12	10 - 20	2.0 - 3.7
Eutrophic	> 12	> 20	< 2.0

Nygaard's phytoplankton community TSIs (Table 2) describe phytoplankton community structures that are indicative of eutrophic and oligotrophic conditions. The scaling of these indices was not conducive to conversion to a Carlson TSI scale and system comparison was therefore done only by general trophic classification. This was also the case with the trophic criteria suggested by USEPA (1975) as supplemental to chlorophyll a concentrations (Table 3). Nygaard's classifications are based on the observations that Cyanophyta, Euglenophyta, centric diatoms, and the Chlorococcales order of Chlorophyta are generally found in waters that are eutrophic. Desmids and penate diatoms, by comparison, generally have a lower tolerance to high nutrient levels (USEPA, 1977).

Table 2.- Nygaard's trophic state indices for phytoplankton communities (described by EPA, 1977).

Index	Oligotrophic	Eutrophic
Myxophyceae / Desmideae	0.0 - 0.4	0.1 - 3.0
Chlorococcales	0.0 - 0.7	0.2 - 9.0
Centric Diatoms / Pennate Diatoms	0.0 - 0.3	0.0 - 1.75
Euglenophyta / Myxophyceae + Chlorococcales	0.0 - 0.2	0.0 - 1.0
Myxophyceae + Chlorococcales + Centric Diatoms + Euglenophyta / Desmideae	0.0 - 1.0	1.2 - 25.0

Table 3.- Supplemental phytoplankton parameters suggested by USEPA (1975) for trophic status classification (described in Armstrong et al, 19887).

Density (No./L)	Oligotrophic	Mesotrophic	Eutrophic
Cell Volume (mm^3/L)	0 - 2,000	2,000 - 15,000	>15,000
Biomass (mg/L)	<5	5 - 30	30 - 100
Primary Prod. (g/m^2-d)	0 - 0.2	0.2 - 0.75	0.75 - 5.0

Table 4.- Comparison of all trophic status indicator classifications for the four impoundments during seasons sampled.

	System / Season						
Indicator	RR.6/89	RR.8/89	RR.12/89	RR.4/90	St.1	St.2	St.3
Chl. <u>a</u>	M	M	O	O	E	E	M
Secchi	E	E	M	E	E	E	O
Phosphorus	E	M	E	E	E	*	*
Phyt.Density	E	E	E	E	E	E	M
Diatom Ratio	E	E	E	E	?	?	?
Cyano. Ratio	E	E	E	E	E	E	E
Euglen.Ratio	?	?	?	?	?	?	?
Compound	E	E	E	E	E	E	E

E = eutrophic M = Mesotrophic O = oligatrophic

* = no data ? = value in both ranges

RR = Ray Roberts Lake St. = Stockpond

Nygaard's phytoplankton community composition had, for the most part, agreement of broad trophic classifications (Table 4). Diatom ratios indicated eutrophic status on all four sampling dates in Ray Roberts Lake. The presence of large numbers of the genus <u>Melosira</u> (a centric diatom) (IAS, 1992) shifted the ratio to an extreme eutrophic classification. The stockponds, by comparison, each had 0 ratio values due to the complete absence of centric diatoms.

As mentioned in the study area description, Stockpond 1 and Stockpond 2 were both visually highly eutrophic. This discrepancy would indicate a limiting factor, possibly silica (necessary for the production of diatom frustules), as well as the potential intolerance to high nutrient levels suggested by Nygaard. Stockpond 3 was not visually eutrophic and also had an absence of centric diatoms. Since Stockpond 2 and 3 were in close proximity to each other, the probability of silica functioning as the limiting factor is supported. The diatom ratio's lack of sensitivity in both Ray Roberts Lake and the three stockponds suggests that this index has little use as a monitoring endpoint by itself. The complete overlapping of value ranges also decreases the applicability of this ratio.

The Cyanophyta/Desmideae ratio (Table 4) showed a similar trend as did the diatom ratio. A dominance of small-celled bluegreen colonies in all Ray Roberts Lake sampling dates (IAS, 1992) shifted this ratio to an extreme eutrophic classification. The three stockponds each had a dominance of bluegreen trichomes which contributed to them being classified as extremely eutrophic. Stockpond 3 did not appear eutrophic, as mentioned before, yet had a high Cyanophyta ratio. This indicates a similar lack of sensitivity and subsequent lack of usefulness of this ratio as a monitoring endpoint.

The Euglenophyta ratio (Table 4) has the same inherent problem as the diatom ratio having completely overlapping ranges. Ray Roberts Lake and all three stockponds had proportionally low Euglenophyta numbers which indicates a lower trophic classification. This is not supported by any of the other indices and does not agree with visual observations of the systems. There is no obvious, or measured, limiting factor that would account for the ratio values. The Euglenophyta ratio does not appear to be applicable as a monitoring tool due to the lack of trophic differentiation, as well as the potential effect of other limiting factors.

The compound ratio, a composite of all the other taxa relative to Desmideae, was more extreme in its eutrophic classifications of each impoundment (Table 4). This was due to the summation of the dominant taxonomic groups compared to a group that never occurred in large numbers (IAS, 1992).

Generally Nygaard's phytoplankton composition indices were not sensitive to differences in the eutrophic and meso-eutrophic classifications which visually appeared to describe the systems studied. A differentiation of oligotrophic from eutrophic might be applicable, but this does not meet the criteria for a useful eutrophication monitoring endpoint mentioned previously. The logic of eutrophic systems being dominated by Cyanophyta, centric diatoms, and members of the Chlorococcales order of Chlorophyta was supported by visual observation and some of the other indices.

The phytoplankton density index, suggested by USEPA (1975) as supplemental to the chlorophyll a index, followed a similar trend as the composition indices. Ray Roberts Lake was classified as eutrophic on each sampling date (Figure 4). The density values, however, did not imply an extreme eutrophic condition. By comparison, both Stockpond 1 and 2 had four times the phytoplankton densities. Stockpond 3 was the only system to fall in the mesotrophic classification range. The classification of each system agrees to a certain degree with the visual observation of each system. Ray Roberts Lake appeared to be moderately eutrophic, Stockponds 1 and 2 highly eutrophic, and Stockpond 3 highly mesotrophic (macrophytes not included). The major weakness of the density index was the lack of sensitivity (resolution) within the broad trophic classifications. As a useful monitoring tool the endpoint needs to be sensitive enough to detect significant change. Conversion of the density index to a Carlson TSI unit based index would improve both the resolution of the index and the comparability to other indices. A potential, and unsubstantiated in this study, weakness is the variability of phytoplankton enumeration and identification stemming from differences in both enumerators and equipment. This index does exhibit agreement, potential sensitivity, and usefulness as a supplemental index (if not a major one).

The chlorophyll a trophic index developed by Carlson (1977) produced values that were in agreement with visual observations and with the general results of the phytoplankton density index. Ray Roberts Lake ranged from a low of 47 units, during December and April sampling, to a high of 54 units, during the August sampling. On a scale of 100 this range does not appear to be of any significance. The NES trophic level classifications place the 47 unit values in the upper end of oligotrophic class and both of the values of 52 and 54 units in the upper mesotrophic class (55 units marking the eutrophic level). Carlson's original breakpoint for the eutrophic class of greater than 50 units placed Ray Roberts Lake in the high mesotrophic to low eutrophic status. Stockponds 1 and 2 had TSI values of 79 and 70 units respectively placing them well into the eutrophic class. Stockpond 3, 51 units, was considered highly mesotrophic by NES classification and low eutrophic by Carlson's classification. The relationship between TSI units, visual observation, and phytoplankton densities appears consistent. The discrepancy appears to be in the differences in the broad trophic classifications. Generally, chlorophyll a measurements do not have the same potential weaknesses observed in phytoplankton enumeration and identification. This indicates that a chlorophyll a TSI would have a sensitivity to change not apparent in the composition and density TSI's.

Secchi disk visibility TSI followed a similar trend as the chlorophyll a TSI (Table 4). The trend was consistently higher for each date and system sampled. Lake Ray Roberts ranged from TSI units of 52 to 65. NES classification placed this from highly mesotrophic to moderately eutrophic while Carlson's demarcations placed the reservoir in the low to moderate eutrophic class. Stockponds 1 and 2 again both fell well into the eutrophic class with unit values of 85 and 73 respectively. Stockpond 3, with 49 units, fell on the NES borderline of oligotrophic and mesotrophic and at the upper mesotrophic range of Carlson. As with the chlorophyll a index, secchi disk visibility TSI appears to exhibit agreement with visual observation and phytoplankton. Measurement sensitivity, not documented here, would also be similar to chlorophyll a. One potential weakness is interference from total suspended solids in the water column, though not a factor in these systems (IAS, 1992). The incorporation of both chlorophyll a and total suspended solids could also be considered a strength of the Secchi disk visibility TSI.

Phosphorus TSI values were more variable than all the other indices considered. Ray Roberts Lake had a TSI unit range from 48 to 72. This range is more extreme than the other unit indices, but more importantly is the lack of 3agreement for the time of occurrence. The low value of 48 units was during the August sampling which was the high point of each of the other indices. The high value of 72 units was during the April sampling which was most often the low point (with December) of the other indices. Phosphorus was determined to be the limiting nutrient in Ray Roberts Lake and therefore not a factor that could account for this dicrepency (Pillard, 1988 and IAS, 1992). Stockpond 1 followed the same trend as the other unit based TSI's with a highly eutrophic 90 units. Data were not available for Stockponds 2 and 3 for this sampling period. Phosphorus (as total phosphorus) measurements do not have a potential sensitivity weakness and a TSI based on this parameter can be an important monitoring tool. The lack of agreement with the other TSI's, however, indicates weakness in the index as a monitoring tool. The form of phosphorus at the time of sampling (ortho- or organic-) may account for this lack of agreement, although many studies have found total phosphorus to be highly correlated with the other trophic parameters.

The relationship between chlorophyll a TSI values and secchi disk visibility TSI values had the best correlation (0.50) of all three relationships measured (Table 5). This relationship explains 50 percent of the variance associated with changes in each parameter. Phosphorus exhibited a weak relationship to both chlorophyll a (r^2 = 0.02) and secchi disk visibility (r^2 = 0.23). The variability of phosphorus concentrations throughout the year was more extreme than the other two parameters. Phosphorus TSI values in Ray Roberts

Lake ranged from 33.2 to 92.3. Chlorophyll a values ranged from 50.4 to 67.7 and secchi disk visibility values ranged from 30.9 to 60.1.

Table 5.- Trophic status indicator value (Carlson TSI units) correlation (r squared) matrix for Ray Roberts, and three stock ponds data.

	Secchi	Phosphorus	Chlorophyll a
Secchi	1.00		
Phosphorus	0.23	1.00	
Chlorophyll a	0.50	0.02	1.00

Conclusions

Chlorophyll a and Secchi disk visibility TSIs had the best level of agreement of all TSIs compared. Phytoplankton density TSI showed agreement with both these TSIs but did not appear as comparable without the prior conversion to TSI units. Each of the phytoplankton community composition TSIs had agreement between themselves but appeared insensitive to differences between systems. Composition TSIs were also influence by factors that are not easily (or usually) monitored, such as silica, and would be impractical to incorporate.

The use of broad trophic classifications was inadequate for both description and monitoring needs. The conversion of parameter values to TSI units, as was done by Carlson, increases the sensitivity and comparability of each index. A standard demarcation of trophic boundaries would be a help in the general communication of a systems trophic condition but does not have the sensitivity necessary for use as a monitoring tool. Of those TSIs compared, only the chlorophyll a and Secchi TSI met the endpoint requirements of social importance, ecological importance, and sensitivity to difference and change. The other TSIs compared appear useful as supplemental indices in their present state. Future benefit may be derived from the increased use of TSI unit based indices as a means to communicate and monitor changes in the trophic status of lentic waters.

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CHAPTER III

VALIDATION OF MODEL PREDICTIONS OF TROPHIC STATUS

Introduction

Increased population with its associated industrial growth, intensification of agricultural production, river-basin development, and recreational use of waters, have made the problem of pollution as complex as it is critical. Human activities, which introduce excess nutrients, along with other pollutants, into lakes, streams, and estuaries, are causing significant change in aquatic environments. The excess nutrients greatly accelerate the process of eutrophication. Accelerated eutrophication causes changes in the flora and fauna by changing the water quality conditions. These changes often interfere with both the use and appreciation of the water. Changes related to excessive nutrient loading are a serious problem in many reservoirs which can result in limits on recreational use and increases in water treatment costs.

A better understanding of the eutrophication process is needed to improve the ability to manage water as a multi-use resource. An understanding of the variability associated with eutrophication indicators, such as chlorophyll a, will improve the ability to monitor water bodies and detect significant changes due to management practices. Successful lake restoration projects such as Medical Lake, Washington (Soltero et al., 1981; Scholz et al., 1985), Lake Washington (Edmunson, 1991), and Shagawa Lake, Minnesota (Larsen et al., 1981) have emphasized the benefits gained from the understanding of the eutrophication process. The National Research Council (1992) stated that there is a need for cost-effective and reliable indicators of ecosystem function that reflect long-term change and response to stress.

A number of trophic status indices and eutrophication models have been developed to describe and predict the trophic status of lakes and reservoirs. An extensive description of the commonly used lake and reservoir models is provided by Svatos (1986) and Armstrong, et al. (1987). Vollenweider (1976) used total phosphorus in conjunction with mean depth and water residence time to predict mean summer epilimnetic chlorophyll a concentration (a trophic status indicator). This model provides the basis for the Organization for Economic Cooperation and Development (OECD) eutrophication model described by Rast and Lee (1983) and Vollenweider and Kerekes (1981). Armstrong, et al. (1987) determined that the OECD eutrophication model to be the most applicable model for prediction of trophic status in Texas reservoirs. Pillard (1988) also determined the OECD model to be the most applicable model to use in conjunction with land use analysis from satellite reflectance data to predict nutrient loadings and chlorophyll a responses in Ray Roberts Lake, Texas following impoundment. The purpose of this paper is to determine the efficacy of the OECD model for predicting the trophic status of a new impoundment.

Study Area

Ray Roberts Lake

Construction of Ray Roberts Lake, under the control of the Army Corps of Engineers, was initiated in September 1981 (USACOE, 1983) and completed in June 1987. The authorized purposes of the project were flood control, water supply, and recreation (USACOE, 1983).

Ray Roberts Lake is located in Denton County and extends into Cooke County in the Elm Fork Arm of the Trinity River and into Grayson County in the Isle du Bois Arm of the Trinity River. The Ray Roberts Dam is an earthen dam located at 96.6 km (river mile 60.0) on the Elm Fork of the Trinity River, 48.3 km (30.0 river miles) upstream of the existing Lewisville Dam, and approximately 16.1 km (10 miles) north of the city of Denton, Texas. The reservoir covers 11,878 hectares (29,350 acres) at the Conservation Pool and 14,933 hectares (36,900 acres) at the Flood Control Pool.

The Elm Fork of the Trinity River, feeding the western arm of Lake Ray Roberts, has a higher annual nutrient load than the Isle du Bois Creek, feeding the eastern arm (Pillard, 1988). Pillard (1988) determined that nutrients in the Isle du Bois Creek is greatly influenced by its watershed, as it has no major point sources. The Elm Fork of the Trinity River, by comparison, has inputs from two wastewater treatment plants and is not watershed influenced to the same degree, although both tributaries have greatest phosphorus and nitrogen concentrations following storm events. Pillard (1988) also determined phosphorus to be the key limiting nutrient in the lake and that the Elm Fork of the Trinity River had a higher concentration of bioavailable phosphorus. IAS (1988) reported higher productivity of both flora and fauna in the Elm Fork compared to the Isle du Bois Creek tributary. IAS (1992) reported similar differences between tributaries two years after impoundment of Ray Roberts Lake. These reports support the basis for Pillard's (1988) predictions (Table 6) of trophic status differences between the two arms of Ray Roberts Lake. Pillard (1988) concluded that:

> "Ray Roberts Lake will be eutrophic. The Trinity arm of the reservoir will display higher eutrophic conditions and will act to trap nutrients before they enter the main reservoir body. The Isle duBois branch will also act as a nutrient sink, although it will

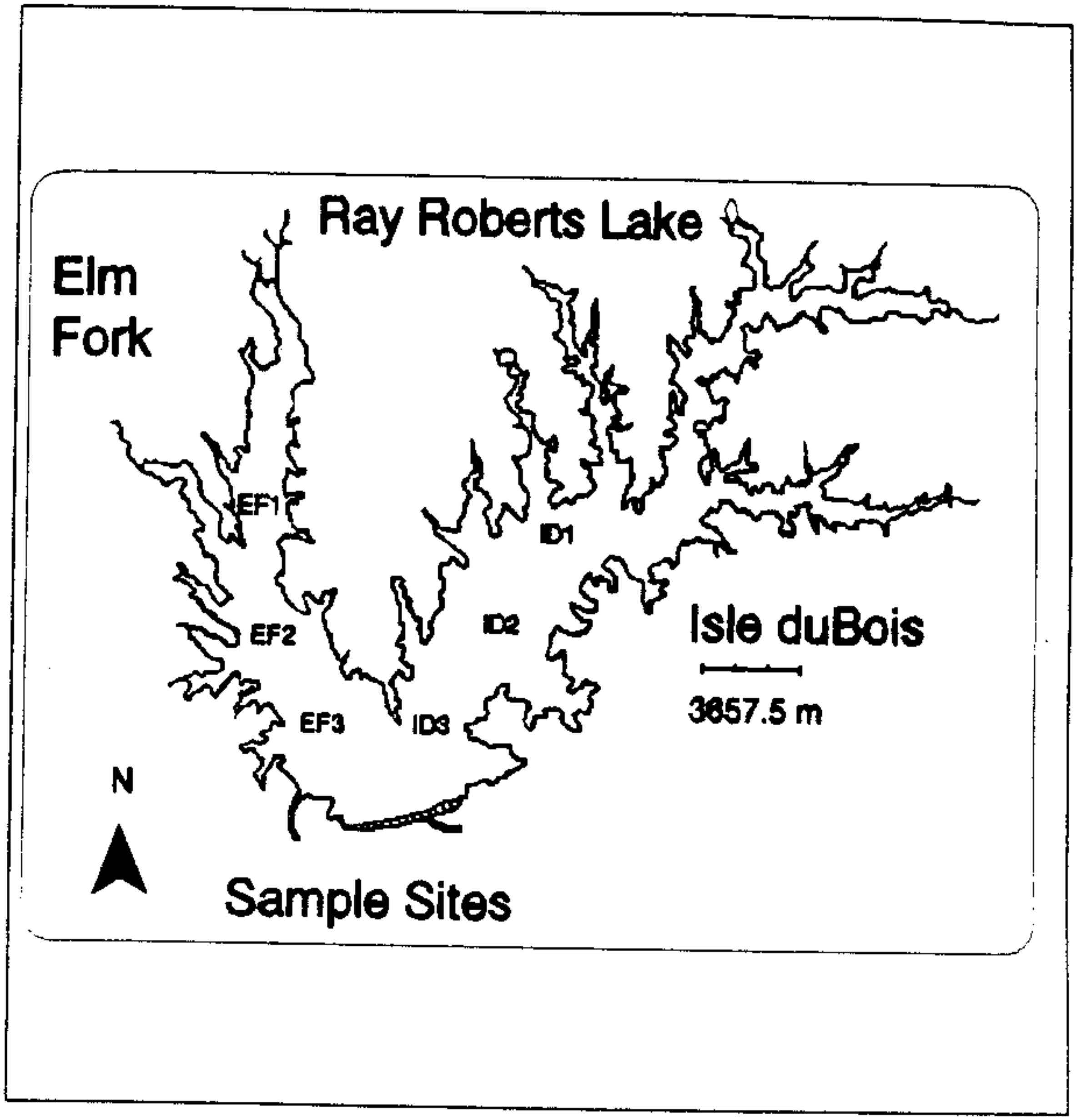
have a lower trophic status and probably fewer eutrophication-related problems."

Table 6. - Predicted minimum, maximum, and mean summer epilimnetic chlorophyll a (ug/L) concentrations for the Elm Fork Arm, Isle du Bois Arm, and Main Body of Ray Roberts Lake.

Area	Minimum	Maximum	Mean
Elm Fork Arm	7.99	102.0	38.0
Isle du Bois Arm	13.00	54.0	29.0
Main Body	14.20	112.0	46.9

* Modified from Pillard (1988)

Map 2 -- Location of Ray Roberts Lake sampling sites for monthly water quality parameters and quarterly plankton parameters from June 1989 to May 1990


Ray Roberts Lake
Elm
Fork
EF1
ID1
ID2
EF2
Isle duBois
3657.5 m
EF3
ID3
N
Sample Sites

Data Collection

Triplicate water samples were collected monthly from six sites in the limnetic, open water, zones of Lake Ray Roberts (Map 2). Sampling began in June 1989 (two years post-impoundment) and continued through May.1990. Subsurface (0.5m) water samples were collected in acid washed two liter Nalgene sampling bottles and placed on ice for laboratory analysis. Samples were analyzed for alkalinity, hardness, chloride, ammonia, nitrate, nitrite, total phosphate, orthophosphate, suspended solids, dissolved solids, chlorophyll a, pheophytin a, turbidity, and sulfate following methods in American Public Health Association (1985). Results of this analysis are reported in IAS (1992). Summer epilimnetic chlorophyll a values from each area of Ray Roberts Lake were compared to predicted values for each area (Table 7).

Additional triplicate surface water samples were collected at 24 sites on Ray Roberts Lake, 7 sites on Lake Kiowa, and 7 sites on Lewisville Lake, on July 5, July 10, July 20, and August 5, 1989 when a potential SPOT (Systeme Pour l'Observation de la Terre) satellite image was taken as part of a congruent study. Water analysis included turbidity, total suspended solids, total dissolved solids, chlorophyll a, and pheophytin a. Chlorophyll a data from Ray Roberts Lake and these two area lakes in the same drainage basin were used to determine chlorophyll a variability.

A statistic that can be used to describe the amount of change in a mean concentration necessary before a new mean is statistically significantly different is the minimum detectable difference (MDD) (Zar, 1984). The value of the MDD is a function of the alpha and beta levels chosen and the variance associated with the original mean estimate. An assumption of the use of the MDD in this study is that as chlorophyll a concentrations increase, variance also increases. Applying this assumption to the prediction of a mean concentration of chlorophyll a implies that as the eutrophication state increases the range associated with the prediction would also increase. For a model to have practical application for monitoring change, the sensitivity of the predicted mean needs to be judged by a consistent measure. The predicted range would also need to have a fine enough resolution to reflect the variability at a given trophic state.

The large number of samples taken during the summer season in the congruent study allows for the calculation of minimum detectable differences (MDD) at a higher degree of probability than with the monthly Ray Roberts Lake sampling data alone. The MDD was used as an aid in the interpretation of the resolution of the chlorophyll a predictions.

Results and Discussion

A similar trend exists between the predicted summer epilimnetic chlorophyll a concentrations and the observed concentrations in the two arms of Ray Roberts Lake two years after impoundment.

Table 7. - Predicted and observed chlorophyll a mean summer epilimnetic concentrations (ug/L) for the main body and each arm (two sites in each) of Ray Roberts Lake.

Area	Minimum		Maximum		Mean	
	Pre.	Obs.	Pre.	Obs.	Pre.	Obs.
Elm Fork Arm	8.0	9.0	102.0	16.0	38.0	12.0
Isle du Bois	13.0	7.0	54.0	20.0	29.0	10.7
Main Body	14.2	6.0	112.0	10.4	46.9	10.0

* Predicted values from Pillard (1988)

The Elm Fork Arm is slightly more eutrophic than the Isle du Bois Arm of Ray Roberts as was predicted by the OECD eutrophication model for the mean summer epilimnetic chlorophyll a concentration. The difference was not statistically significantly different however. Variability was highest in the Isle du Bois Arm. This supports Pillard's conclusion that nutrients from this tributary were more watershed influenced than the Elm Fork which has a greater proportion of point sources. Minimum predicted values were closest to the observed values while mean and maximum values were very different. The observed mean values fell below the predicted range for Ray Roberts Lake in all three areas. This relationship indicates that the OECD eutrophication model was not accounting for one or more limiting factors within the lentic system. The upper sections of the Elm Fork and Isle du Bois Arms, for example, might be acting as nutrient sinks to a greater degree than was expected or decomposition of inundated vegetation might be at a lower than expected rate. Changes in nutrient inputs, scale and sampling site location, and model resolution could also be contributing to the observed difference.

Since Ray Roberts Lake was not completely full during this study period, sample sites EF1 and ID1 were located as close to the tributary inputs as possible without being noticeably influenced by tributary turbidity. A water quality study conducted on Ray Roberts Lake in 1989 reported no significant differences between the two arms in any water quality parameters during the 1989 summer although the trend of higher productivity in the Elm Fork Arm was consistent (IAS (1992). The tributaries Elm Fork and Timber Creek (Map 2) maintained the same differences observed by Pillard (1988). Nitrate, total phosphorus, and orthophosphate were consistently higher in the Elm Fork than in Timber Creek which feeds the Isle du Bois Arm. These differences in nutrient concentrations were not reflected in significantly different chlorophyll a concentrations in the respective arms of Ray Roberts Lake. The difference between predicted and observed mean summer epilimnetic chlorophyll a concentrations in Ray Roberts Lake does not appear to be due to changes in nutrient inputs from the sampled tributaries.

Questions of scale and sample site location were addressed by expanding sampling sites to 24 locations on Ray Roberts Lake, 7 on Kiowa Lake, and 7 on Lewisville Lake on July 5, 10, 20, and August 5. Mean epilimnetic chlorophyll a concentrations for Ray Roberts Lake was not significantly different from the other data sets. Only Lewisville Lake showed a significant change in concentration in August, and this increase was not greater than the 25 ug/L that OECD uses to delineate eutrophic systems (Table 8). All three lakes were similar in chlorophyll a concentrations when considered as a system. This indicates the consistency of the trophic state within the Elm Fork of the Trinity River drainage. The grand mean of all three lakes fell within or just below the predicted range for Ray Roberts Lake's mean summer chlorophyll a concentration. As mentioned before, this consistent underestimation implies an unaddressed, or under weighted, limiting factor in either the nutrient loading model used by Pillard (1988) or in the subsequent OECD chlorophyll a prediction model.

Table 8.- Mean and grand mean epilimnetic chlorophyll a concentrations (ug/L) for Ray Roberts, Lewisville, and Kiowa Lakes on July 5, July 10, July 20, and August 5,1989.

	Ray Roberts	Lewisville	Kiowa
July 5	12.5	11.5	8.8
July 10	14.9	17.2	12.2
July 20	8.3	16.6	16.8
August 5	11.9	23.9	18.7
Grand Mean	11.9	17.3	14.1

Another factor to consider is evaluating the differences between the observed and the resolution or sensitivity of the final prediction. It is possible that the observed chlorophyll a concentrations falling within (or close to) a predicted range means that the OECD model is sensitive and functional for any or all desired applications. Since most models, including the OECD Eutrophication model, determine resolution and sensitivity through minimum and maximum values to derive a range; the physical, chemical, and biological factors which are potential limiting factors are not included in the mathematics of the model. The results of this study indicates that Ray Roberts Lake, as well as the other sampled lakes in the Elm Fork drainage area, have limiting factors functioning that keep chlorophyll a concentrations from increasing much beyond the minimum predicted end of the range. Likely factor candidates are Ray Roberts Lake's function as a nutrient sink and lower than predicted nutrient loading.

Minimum detectable difference (MDD), as mentioned in the methods, describes the amount of difference necessary for significance within set probabilities at different variance levels. The MDD calculated for Ray Roberts Lake, Lewisville Lake, and Kiowa Lake on the four dates in July and August are given in Table 9 with their associated means. The MDD for all three lakes consistently indicates that a 20 to 25 percent difference is necessary to be able to detect a significant change. The predictions of Pillard (1988) of the trophic status for Ray Roberts Lake using the OECD eutrophication model are not close enough to the observed to be considered as accurate predictors of chlorophyll a levels two years post impoundment. Predicted mean summer epilimnetic chlorophyll a concentration of 46.9 ug/L (Table 7) is significantly different from the observed 10.4 ug/L (Table 7) using the MDD measure. A prediction range for Ray Roberts Lake, and the other area lakes as well, would have to have apporximately 5 to 16 ug/L different from the mean to be considered accurate predictors of chlorophyll a concentrations. Changes in this system would have to exceed this range to be considered significant in terms of the MDD

Table 9. - Minimum Detectable Difference and mean chlorophyll a concentrations (ug/L) for Ray Roberts, Lewisville, and Kiowa Lakes on July 5, July 10, July 20, and August 5, 1989.

	Ray Roberts		Lewisville		Kiowa	
	Mean	MDD	Mean	MDD	Mean	MDD
July 5	12.5	2.3	11.5	3.8	8.8	2.0
July 10	14.9	2.4	17.2	1.6	12.2	2.6
July 20	8.3	1.4	16.6	4.3	16.8	2.0
August 5	11.9	2.3	23.9	6.0	18.7	2.0

MDD - alpha = 0.05 and beta = 0.1

for chlorophyll a concentration in a lake of this trophic state. This range is still large considering the differences in trophic status it covers, almost the entire mesotrophic delineation. The predicted mean of 46.9 ug/L places this system well into the eutrophic level using either Carlson's index (Carlson, 1977) or the 20 ug/L eutrophic criterion of OECD.

The resolution of the model itself can be judged similarly. A predicted mean concentration of 46.9 ug/L could be considered successful if observed values fell within the MDD range for that mean concentration, which in this case, is an approximate range between 23 and 70 ug/L. Although this would still mean observed mean values were significantly different from predicted values, the range is more realistic for this predicted mean than the high-low prediction of 13 to 112 ug/L which covers all of the eutrophic and half the mesotrophic trophic levels.

Conclusions

The chlorophyll a predictions of Pillard (1988), using both nutrient loading and OECD eutrophication models, were significantly higher than measured concentrations two years after the impoundment of Ray Roberts Lake. Other factors seem to be limiting chlorophyll a concentrations well before phosphorus becomes the limiting nutrient. This study could not determine which of the model inputs was responsible for the final over estimation or if this was a function of a lack of stabilization of this reservoir. Future summer epilimnetic measurements, as is planned for the sixth year post-impoundment period, will aid in identifying reasons for the observed discrepancies. A major weakness in the modeling approach is the prediction range derived from minimum and maximum value inputs. The model range reported by Pillard (1988) appears too large to be of practical use in both monitoring change and verifying the mean prediction. The development of a consistent measure with which to judge predictions and determine significant change is needed. The minimum detectable difference (MDD) appears to be such a tool with the potential for determining a significant range, as well as, for monitoring future changes in chlorophyll a concentrations. Further development of this tool to encompass a greater range of mean concentrations is needed before the relationship with expected variance can be fully confirmed.

There is a difference in the predicted mean summer epilimnetic chlorophyll a concentrations and the measured concentrations two years post-impoundment. Although the expected differences between the Elm Fork arm, the Isle du Bois arm, and the main body of the reservoir were evident two years after impoundment, the mean predictions were consistently higher than the measured chlorophyll a concentrations.

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CHAPTER IV

MONITORING SELECTED TROPHIC INDICATORS USING REMOTE SENSING

Introduction

Introduction of excess nutrients, along with other pollutants, into lakes, streams, and estuaries, are causing significant change in aquatic environments. The excess nutrients greatly accelerate the process of eutrophication and cause changes in the flora and fauna of the system by changing the water quality conditions. These changes often interfere with both the uses and asthetics of the system. Changes related to excessive nutrient loading are a serious problem in many reservoirs which can result in limits on recreational use and increases in water treatment costs.

Understanding the eutrophication process has improved the ability to manage water as a multi-use resource. An understanding of the variability of eutrophication endpoints will improve the ability to monitor water bodies and detect significant changes due to management practices. Successful lake restoration projects such as Medical Lake, Washington (Soltero et al., 1981; Scholz et al., 1985), Lake Washington (Edmunson, 1991), and Shagawa Lake, Minnesota (Larsen et al., 1981) have emphasized the benefits gained from the understanding of the eutrophication process. There is a continuing need to determine effective endpoints (indicators) to measure or predict which aid in the monitoring of the eutrophication process. The National Research Council (1992) concluded that one area of research needed for lake restoration was:

> "Improved assessment programs are needed to determine the severity and extent of damage in lakes and their change in status over time. Innovative basic research is required to improve the science of assessment and monitoring. There is a great need for cost-effective, reliable indicators of ecosystem function, including those that will reflect long-term change and response to stress. Research on indicators should include traditional community and ecosystem measurements, paleoecological trend assessments, and remote sensing."

The use of remote sensing systems to monitor both land use and water quality has greatly increased over the past 15 years. Remote sensing systems measure reflectance and emmisivity as it varies with spectral wavelength. Statistical analysis of this information can be used for the quantification of land use or any water parameter that influences light reflectance or backscattering. Richie et al. (1987) determined that Landsat Multispectral Scanner data can be effectively used to estimate suspended sediments in aquatic systems where the mean annual concentrations were greater than 5.0 mg/L. Lathrop and Lillesand (1986) found significant relationships between Thematic Mapper (TM) data and Secchi disk depth, chlorophyll a concentrations, turbidity, and surface temperature in Green Bay and Central Lake Michigan. In 1989 Lathrop and Lillesand also found significant correlations of SPOT-1 multispectral data with the same water quality parameters, with the exception of temperature which can not be measured by the Systeme Pour l'Observation de la Terra (SPOT) -1 system. Both Lillesand et al. (1983), using Landsat data, and Wezernak et al. (1976), using low altitude M-7 multispectral scanner, determined that remotely sensed data could effectively predict trophic status of inland water bodies.

The objective of this study was to determine the efficacy of using remotely sensed reflectance data for monitoring trophic status of reservoirs and lakes. Emphasis was placed on determining the lowest magnitude of difference in reflectance values required for statistically different values in water quality. To meet this objective, selected trophic indicator data were collected from three reservoirs in one watershed of North Texas.

Study Areas

Ray Roberts Lake

Construction of Ray Roberts Lake, under the control of the Army Corps of Engineers, was initiated in 1981 and completed in 1987 (USACOE, 1983). Ray Roberts Lake is located in Denton County, extends into Cooke County in the Elm Fork Arm of the Trinity River and into Grayson County in the Isle du Bois Arm of the Trinity River. The Ray Roberts Dam is an earthen dam located 96.6 km (river mile 60.0) on the Elm Fork of the Trinity River, 48.3 km (30.0 river miles) upstream of the existing Lewisville Dam, and approximately 16.1 km (10 miles) north of the city of Denton. The reservoir covers 11,878 hectares (29,350 acres) at the Conservation Pool and 14,933 hectares (36,900 acres) at the Flood Control Pool (Map 3).

Lake Kiowa

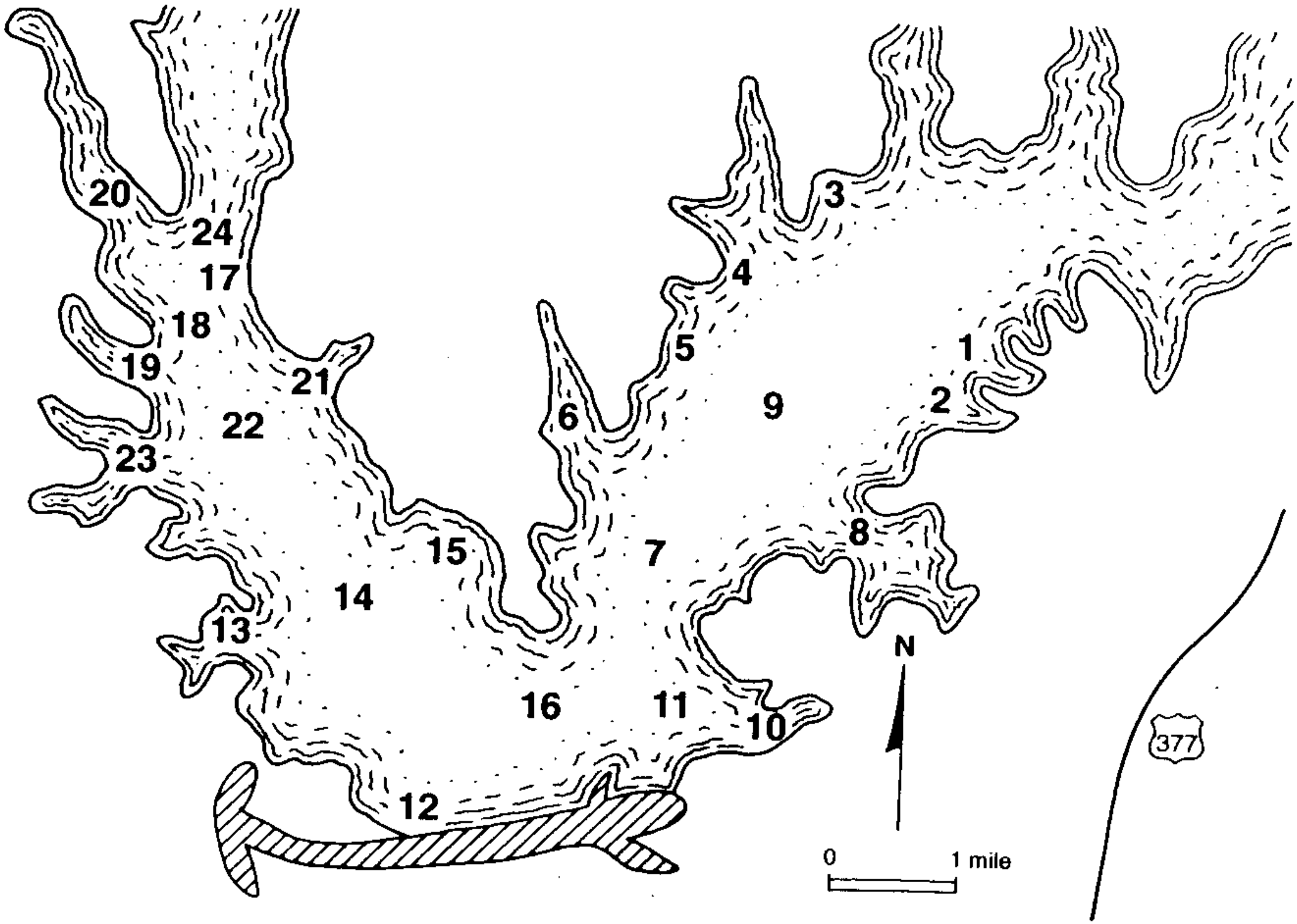
Lake Kiowa is a small private community-owned reservoir located in Cooke County approximately 16.1 km (10 miles) north of Ray Roberts Lake and approximately 32.2 km (20 miles) north of the city of Denton. Surface area at full pool is approximately 226 hectares (560 acres). The north-south fetch is approximately 8.1 miles long. The fixed spillway limits maximum depth to 30 feet. Permission and information was provided courtesy of Lake Kiowa Property

Owners, Assn. This reservoir is part of the Ray Roberts Lake drainage area imputing the Isle du Bois Arm (Map 4).

Lewisville Lake

Lewisville Lake is a reservoir controlled by the Army Corps of Engineers located in Denton County on the Elm Fork of the Trinity River. Impoundment of Lewisville Lake was initiated in 1954 and achieved full capacity in 1957. The lake covers 94.2 square kilometers (23,280 acres). Prior to the completion of Lewisville Lake, Lake Dallas (about 35% of the Lewisville surface area) existed in what is now the Elm Fork Arm of Lewisville Lake (TRA, 1976). The earthen dam is located at river kilometer 44 (Map 4). Since only the upper portions (northern) of Lewisville Lake were visible in the satellite image, only these areas were sampled (Map 5).

Map 3 -- Location of Ray Roberts Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.



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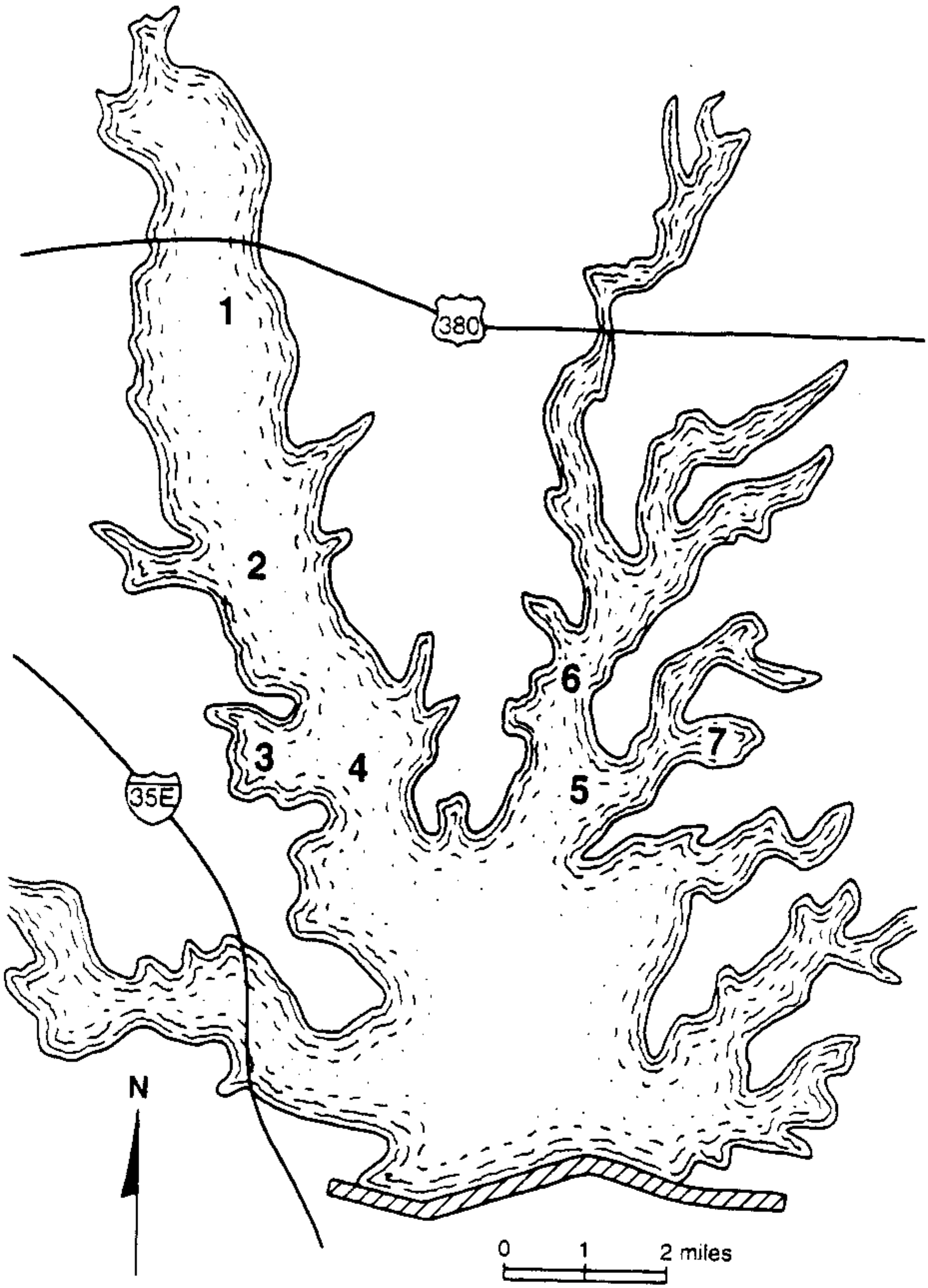
Map 4 -- Location of Kiowa Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.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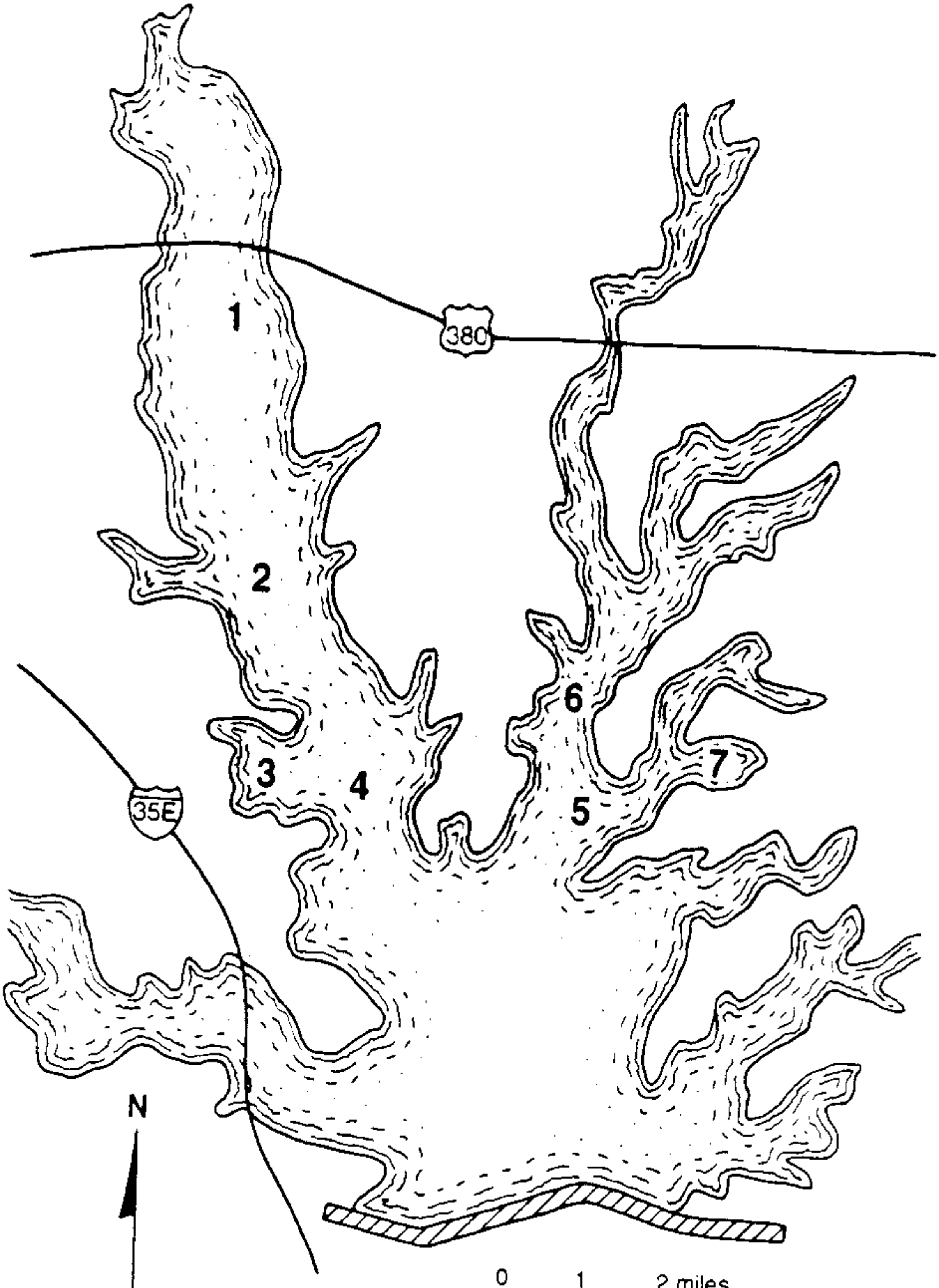
Map 5 -- Location of Lewisville Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.



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Methods

Triplicate surface water samples were collected at 24 sites on Ray Roberts Lake, 7 sites on Lake Kiowa, and 7 sites on Lewisville Lake (Maps 3, 4, and 5). Collections were made on July 5, July 10, July 20, and August 5, 1989 when a potential SPOT (Systeme Pour l'Observation de la Terre) satellite image could be taken. Only the July 20, 1989 image was obtained due to excessive cloud cover on the other dates. Lakes and sites were selected to be within one satellite image. Surface water samples were collected concurrently with time of satellite overpass (plus or minus 0.5 hours). Water samples were placed on ice in the field and returned to the laboratory for analysis. Water clarity analysis included turbidity, total suspended solids, total dissolved solids, chlorophyll a, and pheophytin a following methods in American Public Health Association (1985).

Reflectance data from the SPOT image were analyzed at the Center for Remote Sensing and Landuse Analysis (CRSLA) at the University of North Texas using Earth Resource Data Analysis System (ERDAS). The mean of a five square pixel group was derived for each of the three wavelength bands at each sampling site to insure encompassing of the site within the pixel group. Multiple regression and canonical correlation were performed on the water parameters and reflectance values for each of the three bands and band ratios.

A statistical tool that can be used to describe the amount of difference due to variability for a mean concentration in a system at a set period of time is the minimum detectable difference (MDD) (Zar, 1984). An assumption of the MDD method as used here is that as chlorophyll a concentrations increase the variance also increases. Applying this assumption to the prediction of a mean concentration of chlorophyll a means that as the eutrophication state increased the range of chlorophyll a associated with the prediction would also increase. For remote sensing to have practical application for monitoring change, the sensitivity of the relationship between reflectence and water quality parameters needs to be judged by a consistent measure. The large number of samples taken during the summer season allows for the calculation of minimum detectable differences (MDD) to be calculated at a higher degree of probability than with the monthly Ray Roberts Lake sampling data alone. The MDD was used as an aid in the interpretation of the resolution of the relationship between chlorophyll a and reflectence. Chlorophyll a data from all four sampling dates were used to calculate the Minimum Detectable Difference (MDD) to determine the degree of sensitivity necessary to detect change. This difference was then used as a measure of the efficacy of remotely sensed data as a monitor of eutrophication in terms of chlorophyll a concentrations.

Results and Discussion

A positive correlation of 0.72 between turbidity (Tur.) and Chlorophyll a (Chl.) is apparent in the data from the three reservoirs (Table 10). This relationship, in conjunction with a weak relationship of turbidity to total suspended solids (0.36), indicates that most of the water turbidity is dependent on the phytoplankton biomass. Log transformations of water parameters did not improve any of the relationships. The lack of improvement was unexpected since Lathrop and Lillesand (1989) found strong correlation (0.942) between these two parameters.

Table 10.- Correlations among the water parameters for Ray Roberts, Lewisville, and Kiowa Lakes on July 20, 1989

	TSS	TDS	Tur.	Chl.	Pheo.
TSS	1.00				
TDS	-0.08	1.00			
Tur.	0.36	0.02	1.00		
Chl.	0.19	0.22	0.72	1.00	
Pheo.	-0.38	0.10	-0.35	-0.38	1.00

Examination of the correlation matrix between band reflectence values shows a high relationship between every one of the bands. These correlations ranged between 0.92 and 0.99 (Table 11). The lack of difference among the bands implies that the green wavelength band (Band 1), red wavelength band (Band 2), and near infrared wavelength band (Band 3) either did not change significantly at any site or changed similarly at all sites. Lathrop and Lillesand (1989) reported high correlation (r^2) between Bands 1 and 2 (0.902), but weak correlation (0.541 and 0.74) for the other two relationships. A significant relationship between Band values and water parameters was not expected due the high correlation of the bands.

Table 11.- Correlations among the band values for Ray Roberts, Lewisville, and Kiowa Lakes on July 20, 1989

	Band 1	Band 2	Band 3
Band 1	1.00		
Band 2	0.92	1.00	
Band 3	0.95	0.99	1.00

The correlation of water parameters and band values (Table 12) shows only weak relationships. Total suspended solids (TSS) and total dissolved solids (TDS) were both completely neutral to changes in all bands ranging from a low correlation of 0.01 to a high of only 0.10. Turbidity had the highest correlation of 0.66 with Band 2, but also similar relationships to Band 1 and 3 (due to the high level of agreement between bands). Chlorophyll a followed the same trend as turbidity with the highest correlation of 0.56 with Band 2. This similar trend was expected since it appeared that the predominant source of turbidity was in the phytoplankton biomass. Moore (1980) found the longer wavelengths comprising Bands 2 and 3 had a high response to phytoplankton and suspended sediments. Lathrop and Lillesand (1989) determined that Bands 2 and 3 exhibit poor correlation with turbidity, less than 4.0 NTU and total suspended solids less than 10 mg/L, but showed a strong response to higher levels. They found that the ratio Band 2/Band 1 was more sensitive to the lower concentrations.

Data from this study have shown that the ratio of Band 2/Band 1 did not improve the relationship with chlorophyll a. Correlation with this ratio did not exceed 0.01 for a positive response to any parameter. Additional combinations of bands were attempted to determine the best relationship. Combinations tried were: Band 2 + 1, Band 2 + 3, Band 1 + 3, Band 1 + 2 + 3, Band 2/1, Band 3/1, and Band 3/2. In no case did correlations improve over the single band relationship.

Canonical correlation analysis of water parameters (Table 13) shows that turbidity and chlorophyll a contribute 0.91 and 0.76 respectively to the changes in the first (best) canonical water variable. This relationship is in agreement with the results of the multiple regression analysis in which turbidity and chlorophyll a had a correlation coefficient of 0.71. The second canonical variable shows a lower relationship between total suspended solids and turbidity with contributions of 0.90 and 0.38 respectively. The third canonical variable

Table 12.- Correlations between the water parameters and the band values for Ray Roberts, Lewisville, and Kiowa Lakes on July 20, 1989

	Band 1	Band 2	Band 3
TSS	0.10	0.01	0.02
TDS	0.10	0.07	0.08
Tur.	0.57	0.66	0.62
Chl.	0.48	0.56	0.54
Phe.	-0.15	-0.19	-0.16

shows the least influential relationship where total dissolved solids and pheophytin concentrations contribute 0.42 and 0.75 to changes in the variable.

Canonical correlation of the band values and the first canonical variable (Table 14) again shows the dominance of Band 2 and the high level of agreement with the other bands.

Table 13.- Canonical correlation between the water parameters and their canonical variables.

	Variable 1	Variable 2	Variable 3
TSS	-0.03	0.90	-0.31
TDS	0.05	0.29	0.42
Tur.	0.91	0.38	-0.01
Chl.	0.76	0.22	0.25
Pheo.	-0.31	-0.13	0.75

The low correlation value of 0.75 highlights this similarity of response with the canonical band variable. The remaining canonical band variables show no relationship of any great degree. These relationships indicate that Band 2 is not only the most influential, but is so similar (in its changes) to the other bands that it must be weighted the highest in subsequent analysis.

Table 14.- Canonical correlations between the band values and their canonical variables.

	Variable 1	Variable 2	Variable 3
Band 1	0.75	0.55	0.37
Band 2	0.93	0.21	0.30
Band 3	0.87	0.24	0.42

Canonical correlations between the water parameters and the first 'M' canonical variable of the band values emphasizes turbidity as the primary parameter for response to changes in band reflectance values. Almost 70 percent of the variance associated with changes in band values can be attributed to turbidity (Table 15). Turbidity, however, is influenced by suspended sediments as well as the plankton densities. Total suspended solids were consistently low in all three lakes averaging only 8.53 mg/L. The correlation between TSS and turbidity was 0.36 indicating that only a small proportion of the turbidity could be attributed to suspended solids. The higher correlation of turbidity with chlorophyll a (0.72) (Table 10) indicates that the suspended solids are likely composed of the solid fraction of the plankton biomass. Even though both canonical and multiple correlations show turbidity to be the most responsive parameter in this study to changes in band values, it still has the potential of being influenced by both suspended sediment and plankton biomass. As a monitoring tool, turbidity has limited useful function. Chlorophyll a, on the other hand, is useful as a monitoring tool because of its social (color of water) and ecological importance, as well as, being the most predicted endpoint of eutrophication endpoints. The sensitivity, or the amount of change in chlorophyll a concentration necessary before statistical significance can be established, remains to be determined.

Chlorophyll a data from all three reservoirs on the four dates were used to calculate the minimum detectable difference (MDD) for each system following equations in Zar (1984) (Table 16). Mean chlorophyll a concentrations were not greatly

different between the three reservoirs or between dates. Minimum detectable differences for chlorophyll a were also similar for all three reservoirs. Considering the July 10 relationship between concentration and MDD, there was not enough difference between the systems to be significant.

Table 15.- Squared multiple correlations between the water parameters and the first 'M' canonical variables of the band values

	M Band Variable 1	M Band Variable 2	M Band Variable 3
TSS	0.00	0.06	0.07
TDS	0.00	0.01	0.07
Tur.	0.69	0.11	-0.02
Chl.	0.58	0.06	0.04
Phe.	-0.23	-0.04	0.13

This indicates that the range of chlorophyll a concentrations measured was not large enough to produce significant differences. This also supports the weak correlation of chlorophyll a and Band 2 (0.56). The correlation implies that a response relationship exists but that this relationship is not significant within the measured range.

Table 16. - Minimum Detectable Difference and mean chlorophyll a concentrations (ug/L) for Ray Roberts, Lewisville, and Kiowa Lakes on July 5, July 10, July 20, and August 5, 1989.

	Ray Roberts		Lewisville		Kiowa	
	Mean	MDD	Mean	MDD	Mean	MDD
July 5	12.53	2.29	11.48	3.79	8.81	1.96
July 10	14.93	2.38	17.22	1.60	12.22	2.56
July 20	8.26	1.42	16.56	4.32	16.83	2.04
August 5	11.86	2.30	23.88	6.01	18.71	1.96

MDD - alpha = 0.05 and beta = 0.1

Conclusions

This study's goal of using remotely sensed reflectance values for monitoring water quality was not realized. The three lakes sampled were not significantly different in the parameters measured. Turbidity in all the systems appears to be primarily due to plankton biomass not suspended sediments. The correlations between turbidity and chlorophyll a with band values is high enough to indicate a relationship exists but too low to be of practical use as a monitoring tool by itself. Band 2 was the most responsive to changes in both chlorophyll a and the resulting turbidity. This supports the results described by Lathrop and Lillesand (1989) where the red and near infrared wavelengths responded best to changes in chlorophyll a concentrations. Band ratioing, however, did not improve response correlations nor were the band correlations as high as Lathrop and Lillesand (1989) data. This lack of similarity was attributed to the difference in the range of parameter values (Table 17).

The magnitude of the range appears to be directly related to the degree of correlation. This relationship is to be expected. The sensitivity of remotely sensed reflectance to changes in water quality parameters (especially chlorophyll a) is not readily apparent in spite of the correlation. This study did not seem to have a range large enough to be significant both in terms of the statistics of the July 10 data and the MDD of all four dates. The studies used above for comparison, on the other hand, had a magnitude large enough to produce a significant correlation but sensitivity is not apparent.

Further studies of these variables will likely improve interpretation of the sensitivity of remotely sensed reflectance data for monitoring water quality parameters. Decreasing sensing distance with boom mounted sensors, as is currently being studied, will improve knowledge of band and band ratio responses. The development of the minimum detectable difference for a monitored parameter, such as chlorophyll a, shows potential for standardizing the determination of prediction sensitivity.

Table 17.- Comparison of parameter range and r^2 with reflectance (in parenthesis) for chlorophyll a, turbidity, and total suspended solids between this study and similar studies.

Study	Chlorophyll a	Turbidity	TSS
This	4.7 - 22.0	4.4 - 12.0	2.0 - 19.0
	(0.41)	(0.54)	(0.23)
1*	12.7 - 76.9	1.2 - 11.9	4.6 - 28.9
	(0.90)	(0.88)	(0.93)
2*	1.0 - 50.3	0.54 - 12.0	<1 - 62.8
	(0.98)	(0.99)	(0.30)
3*	0.88 - 25.3	NA	NA
	(0.74)		

1* = Lathrop and Lillesand (1989)

2* = Lathrop and Lillesand (1983)

3* = Verdes (1985)

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CHAPTER V

DEVELOPMENT OF A CHLOROPHYLL *a* VARIABILITY MODEL

Introduction

Successful lake restoration projects such as Medical Lake, Washington (Soltero *et al.*, 1981; Scholz *et al.*, 1985), Lake Washington (Edmunson, 1991), and Shagawa Lake, Minnesota (Larsen *et al.*, 1981) have shown that the success of such projects is based on understanding the eutrophication process. A better understanding of the eutrophication process will improve the ability to manage water as a multi-use resource. Understanding the variability of eutrophication endpoints will improve the ability to monitor water bodies and detect significant changes due to management practices.

Generally the trophic status of lakes and reservoirs is based on the nutrient (usually phosphorus and nitrogen) concentrations and the associated phytoplankton productivity. Changes in trophic status are often related to induced increases or decreases in the allocthonus inputs of nitrogen and phosphorus. Lake and reservoir managers often have limited data sets for analysis of changes in the trophic status of a lentic ecosystem. A common property of limited data sets is elevated variance. As lake managers try to quantitatively evaluate the success or failure of a management strategy, the magnitude of the change necessary for statistical significance is often very large. In those cases where demonstrating statistically significant change is important, the analysis presented below may be useful.

The variance associated with an endpoint in many eutrophication models (Armstrong et al, 1987) is rarely addressed. This variance, however, is key for the determination of significant difference and/or successful restoration. Peterson (1990) provides a strong rationale for the need to address both the level (alpha) and power (beta) in ecological monitoring and management by showing the danger of committing either a Type I or a Type II error. To assist managers, a need exists for a consistent measure of the variability associated with the increasing levels of trophic status endpoints. A model developed to fill this need should be statistically based, reflect parameter variability, be applicable at different scales, and be both ecologically and socially relevant.

Chlorophyll _a_ seems to be the best endpoint used in eutrophication studies on which to develop a measure of variance for the following reasons:

1. Chlorophyll _a_ is a commonly predicted endpoint of trophic status models (Armstrong et al, 1987).
2. Chlorophyll _a_ reflects phytoplankton biomass,

the predominant source of turbidity in the pelagic zones of many lakes and reservoirs.

3. Chlorophyll a measurements are cheaper to obtain in large numbers than nutrient measurements.

4. Phytoplankton, with its associated turbidity, color, taste and odor is of great social importance.

A statistic that can be used to describe the amount of change in a mean concentration necessary before a new mean is statistically significantly different is the minimum detectable difference (MDD) (Zar, 1984). The value of the MDD is a function of the alpha and beta levels chosen and the variance associated with the original mean estimate. An assumption of the use of the MDD in this study is that as chlorophyll a concentrations increase, variance also increases. Applying this assumption to the prediction of a mean concentration of chlorophyll a implies that as the eutrophication state increases the range associated with the prediction would also increase. For a model of the type developed here to have practical application for monitoring change, the sensitivity of the predicted mean needs to be judged by a consistent measure. The predicted range would also need to have a fine enough resolution to reflect the variability at a given trophic state.

The Minimum Detectable Difference (MDD) was chosen as a measure of variability because:

1. MDD is a statistic which incorporates variance and the mean as does the coefficient of variance, used to describe sample variance.
2. Unlike the coefficient of variance, MDD includes calculation provisions for incorporating a measure of level (alpha) and power (beta).
3. The alpha and beta probabilities can be set at a consistent and predetermined level.
4. MDD can be applied to different scales (site, system, ecosystem).
5. The confidence limits about the regression line of a MDD for a trophic parameter like mean chlorophyll a concentration is adaptable for the development of a range (window) usable as a consistent aid in the interpretation of model predictions, parameter changes, and statistical differences.

The objective of this study was to develop a statistically based tool that would aid lake and reservoir managers in determining how much change in chlorophyll a concentrations would have to occur before the change would be deemed statistically significant. The statistical significance of a change in a system (whether negative or positive) depends in part on the variance associated with the data analyzed. The objective in model development was to quantify the relationship of variance associated with a range of chlorophyll a concentrations with specific or predetermined statistical probability. Successful development of the MDD model will provide a statistically based tool useful to water resource managers in assessment of eutrophication impact, restoration changes, long term monitoring, and prediction ranges.

Study Areas

To meet the objective set out above, chlorophyll a data from six lakes were selected for preliminary model development. These systems had chlorophyll a concentrations ranging from about 10 ug/L (mesotrophic) to over 100 ug/L (hyper-eutrophic). This range encompasses the trophic levels of greatest concern for restoration and monitoring. The size of each system varied greatly and they were all located in the state of Texas. These lakes were chosen for the consistency of sampling and measurement, all being done by the Institute of Applied Sciences at the University of North Texas.

Ray Roberts Lake

Construction of Ray Roberts Lake, by the Army Corps of Engineers, was initiated in 1981 and completed in 1987 (USACOE, 1983). Ray Roberts Lake is located in Denton County extends into Cooke County in the Elm Fork arm of the Trinity River and into Grayson County in the Isle du Bois arm of the Trinity River. The Ray Roberts Dam is an earthen dam located 96.6 km (river mile 60.0) on the Elm Fork of the Trinity River, 48.3 km (30.0 river miles) upstream of the existing Lewisville Dam, and approximately 16.1 km (10 miles) north of the city of Denton. The reservoir covers 11,878 hectares (29,350 acres) at the conservation pool and 14,933 hectares (36,900 acres) at the flood control pool (Map 6). Ray Roberts Lake appears to be a mesotrophic to low eutrophic reservoir (IAS, 1992).

Lake Kiowa

Lake Kiowa is a small private community owned reservoir located in Cooke County approximately 16.1 km (10 miles) north of Ray Roberts Lake and approximately 32.2 km (20 miles) north of the city of Denton. Surface area at full pool is approximately 226 hectares (560 acres). The north-south fetch is approximately 8.1 miles long. The fixed spillway limits maximum depth to 30 feet. Permission to sample the lake and information about the lake was provided courtesy of Lake Kiowa Property Owners, Assn. This reservoir is part of the Ray Roberts Lake drainage area in the Isle du Bois Arm (Map 7). Lake Kiowa appears to be a low eutrophic lake from visual observation.

Lewisville Lake

Lewisville Lake is an Army Corps of Engineers reservoir located in Denton County on the Elm Fork of the Trinity River. Impoundment of Lewisville Lake was initiated in 1954. Full capacity was achieved in 1957. The lake covers 94.2 square kilometers (23,280 acres). Prior to the completion of Lewisville Lake, Lake Dallas (about 35% of the Lewisville surface area) existed in what is now the Elm Fork arm of Lewisville Lake (TRA, 1976). The earthen dam is located at river kilometer 44 (Map 8). Lewisville Lake visually appears to be a mesotrophic to low eutrophic reservoir.

Stockpond 1

Stockpond 1 is a tertiary lagoon system (approximately 5 acres) for an industrial plant on the Texas coast. This pond is a hypereutrophic system. The water in the last section of the tertiary lagoon and the settling pond is "pea soup" green from planktonic algae. The littoral zone has productive populations of shrimp and aquatic snails. The shoreline also has large numbers of feeding egrets and herons. The primary purpose of the lagoon system is the physical, chemical, and biological treatment of the effluent from the industrial plant.

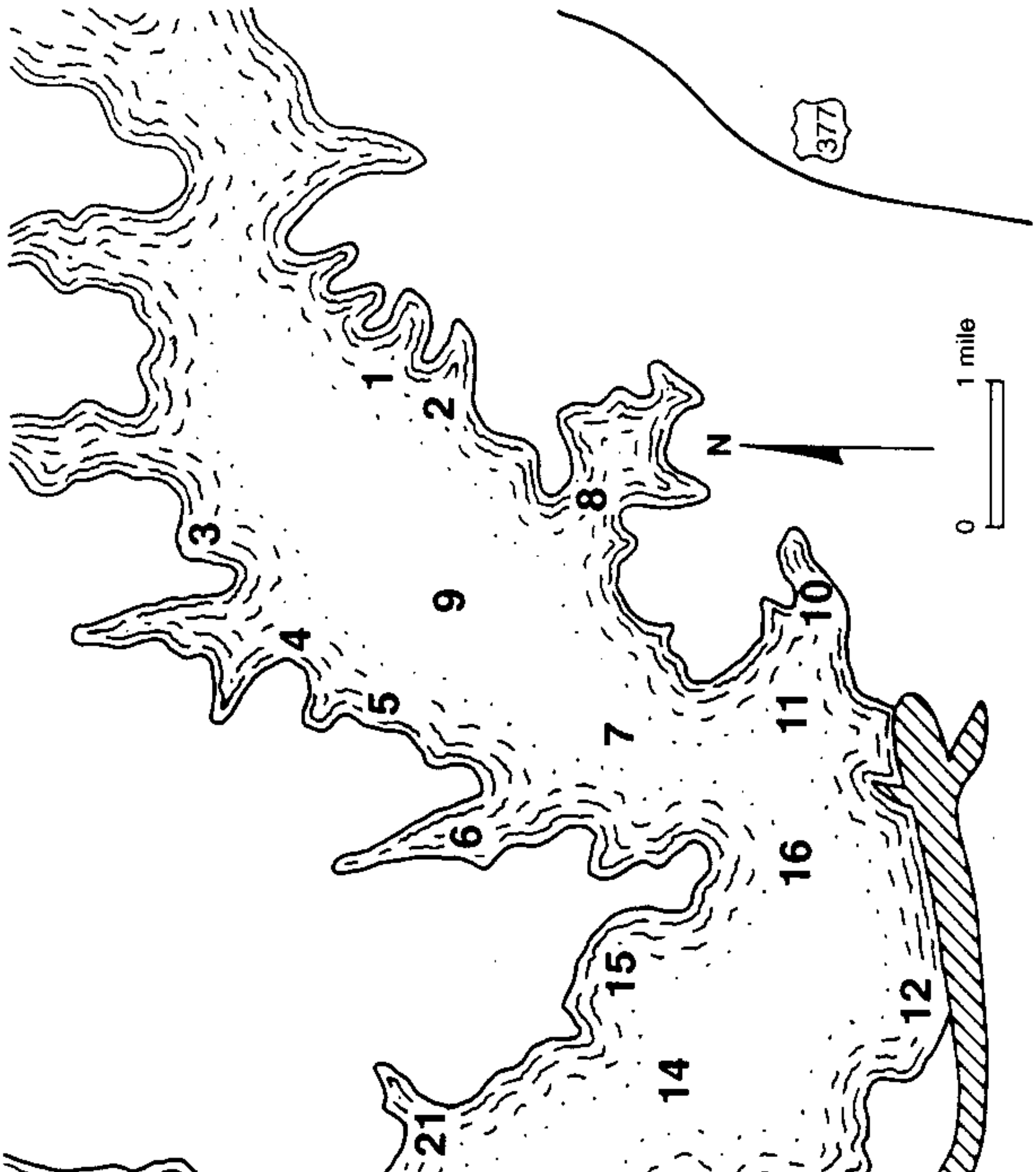
Stockpond 2

Stockpond 2 is a small earthen dam impoundment (approximately 0.5 acres) located in Southeast Denton County, Texas. Stockpond 2 is spring fed and receives runoff from the lawns around a large house located within 100 yards of the pond. Visually the pond appears to be highly eutrophic with dense mats of filamentous algae in the littoral zone. Anoxic odors are also present.

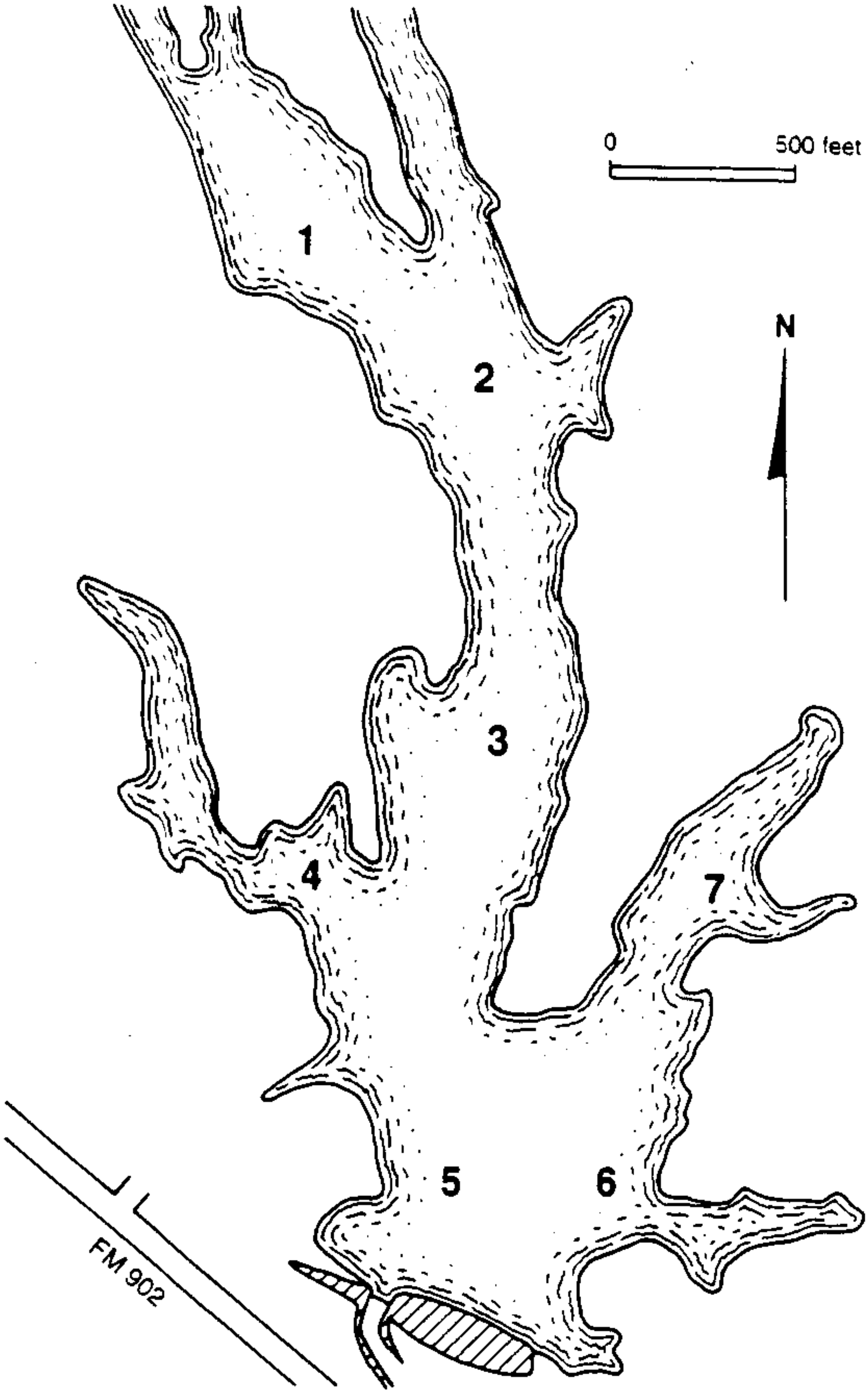
Stockpond 3

Stockpond 3 is a small earthen dam impoundment (approximately 0.5 acres) located near Stockpond 2. The drainage area for Stockpond 3 is separated from Stockpond 2 drainage area by a ridge. Stockpond 3 receives water only from ground water infiltration and surface runoff from a predominantly timbered watershed. This pond appears much less eutrophic although the littoral zone is dominated by macrophytes.

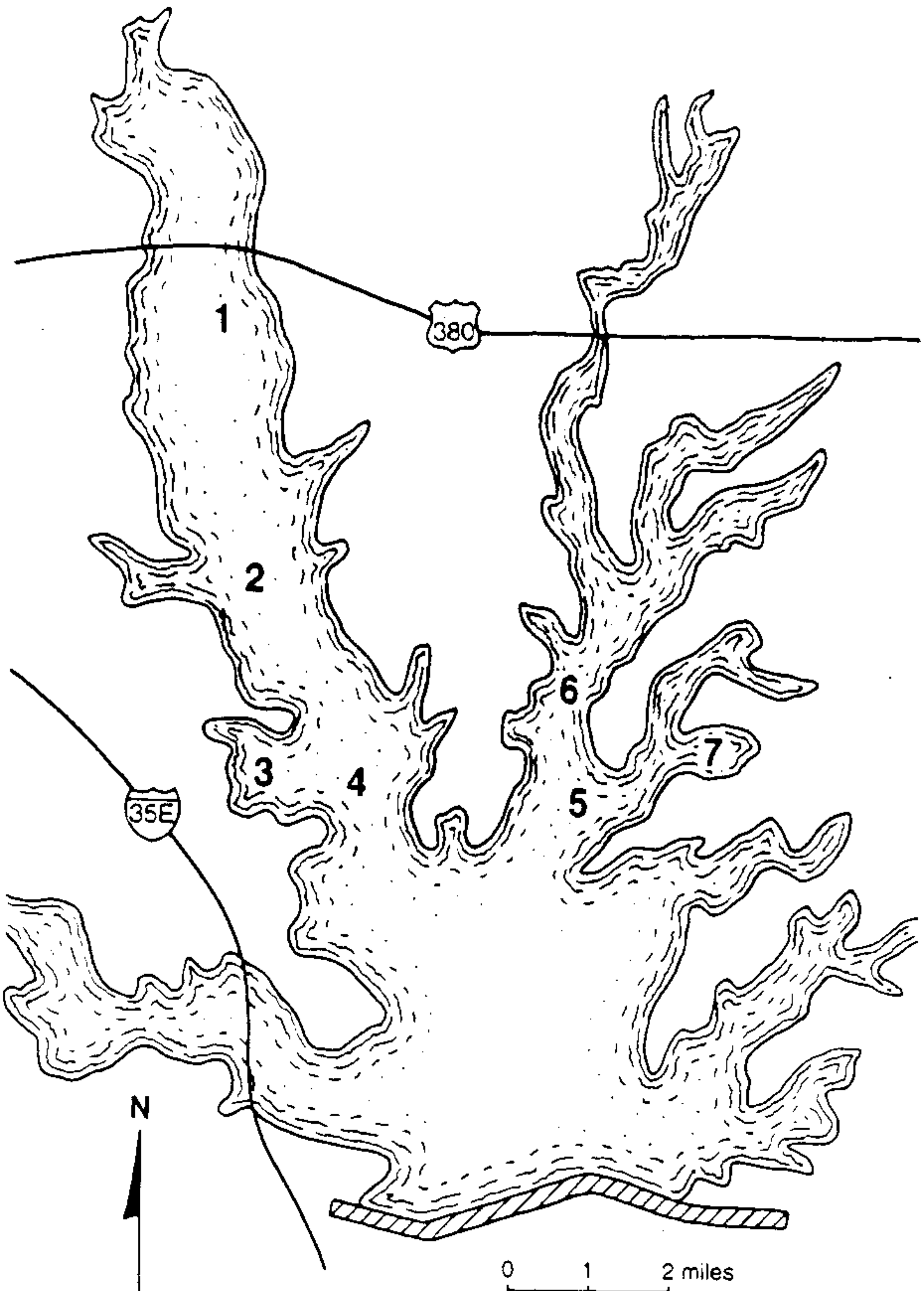
Map 6 -- Location of Ray Roberts Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.


377
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
N
0
1 mile

Map 7 -- Location of Kiowa Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.


0
500 feet
N
1
2
3
4
5
6
7
FM 902

Map 8 -- Location of Lewisville Lake sampling sites for water clarity parameters and remotely sensed reflectences on July 5, July 10, July 20, and August 5, 1989.


1
2
3
4
5
6
7
380
35E
N
0
1
2 miles

Data Collection

Triplicate water samples were collected monthly from six sites in the limnetic, open water, zones of Lake Ray Roberts (Map 5). Sampling began in June 1989 and continued through May 1990. Subsurface (0.5 m) water samples were collected in acid washed two liter Nalgene sampling bottles and placed on ice for laboratory analysis. Samples were analyzed for alkalinity, hardness, chloride, ammonia, nitrate, nitrite, total phosphate, orthophosphate, suspended solids, dissolved solids, chlorophyll a, pheophytin a, turbidity, and sulfate following methods in American Public Health Association (1985). Results of these analyses are reported in IAS (1992).

Additional triplicate surface water samples were collected at 24 sites on Ray Roberts Lake, 7 sites on Lake Kiowa, and 7 sites on Lewisville Lake. Sample dates were on July 5, July 10, July 20, and August 5, 1989 when a SPOT (Systeme Pour l'Observation de la Terre) satellite image might be taken as part of a congruent study for the use of remotely sensed data to assess trophic status. Water analysis included turbidity, total suspended solids, total dissolved solids, chlorophyll a, and pheophytin a.

Stockpond 1 was sampled on three successive days (September 28, 29 and 30, 1990). Stockponds 2 and 3 were sampled on August 16, 1991. Water analysis included all the parameters described for the Ray Roberts Lake monthly sampling.

Model Development

The Minimum Detectable Difference (MDD) may be calculated by different methods depending on the type of data base and the objectives for use (Zar, 1984). A MDD can be calculated, for example, to test for the smallest detectable difference between two existing population means. The purpose in this study, however, was to detect the difference between an existing mean and the mean of the same population at a future time. The MDD calculation used for this purpose was:

$$G = s^2/n \; (t_{\alpha,v} + t_{\beta(1),v})$$

G = Minimum Detectable Difference

s^2 = variance

n = sample size

α = level of significance

v = degrees of freedom

β = power of the statistic

This equation allows for setting both the level of significance (α) and the power of the test (β), in the example presented here $\alpha = 0.05$ and $\beta = 0.1$ respectively. The incorporation and standardizing of the probability of committing both Type I and Type II errors was felt to be important in the development and potential use of any new statistical application. The weakness was limited to different sample sizes and their associated degrees of freedom. The MDD model development assumes that all samples were from pelagic zones with minimal littoral influence, such as periphyton and macrophytes.

The MDD equation also has the option of one-tailed or two-tailed calculation. One-tailed derived MDD and two-tailed derived MDD are different only in their degrees of sensitivity and in the question they are meant to answer. Two-tailed MDD calculation would increase the width of the 95% confidence band by increasing the alpha value (Beta remains constant). This model is a representation of increasing variability with increasing chlorophyll a concentration which may change in either direction. The model derived from a MDD calculated one-tailed is appropriate when answering a directional question such as whether chlorophyll a concentrations improved (lessened) after management practices. The width of the 95% confidence band would be narrower and, therefore, more sensitive to smaller changes. Since the ultimate benefit of the model is its usefulness for detecting significant differences in response to directional management practices, the MDD was calculated as one-tailed.

The 95 percent confidence bands (Window) were calculated following Zar (1984) for predicting the mean value of Y (MDD) in a population at a given level of X (log chlorophyll a concentration). Probability values were set at the 0.05 level and two-tailed, with only the standard error of the Y and the degrees of freedom influencing the width of the band. The resulting area between the 95 percent bands would be applicable to answer the questions of whether a system was restored by a significant amount or whether a system was impacted by a significant amount. Applications of the MDD model (whether impact or restoration) require only the upper 95 percent confidence limit as only the greatest amount of variability for significance is appropriate.

Results

MDD models were calculated with different temporal combinations to help determine the influence of temporal scale on the sensitivity and practical use of the resulting window. As mentioned previously, the question of spatial scale was not addressed directly since the number of samples was small (three) at each individual site. An assumption was made that the relationship between increasing variability and increasing chlorophyll a concentration would be maintained regardless of the spatial scale. The correctness of this assumption will have to be tested. Temporal scale was considered by treating each sampling day during the summer as a separate system, combining two days within each lake as a separate system, and finally combining all sample days of summer within each lake as a separate system. The last combination is logically similar to the mean summer epilimnetic chlorophyll a concentration used by the Organization for Economic Cooperation and Development (OECD) as a predicted endpoint for its eutrophication model (Armstrong et al, 1987).

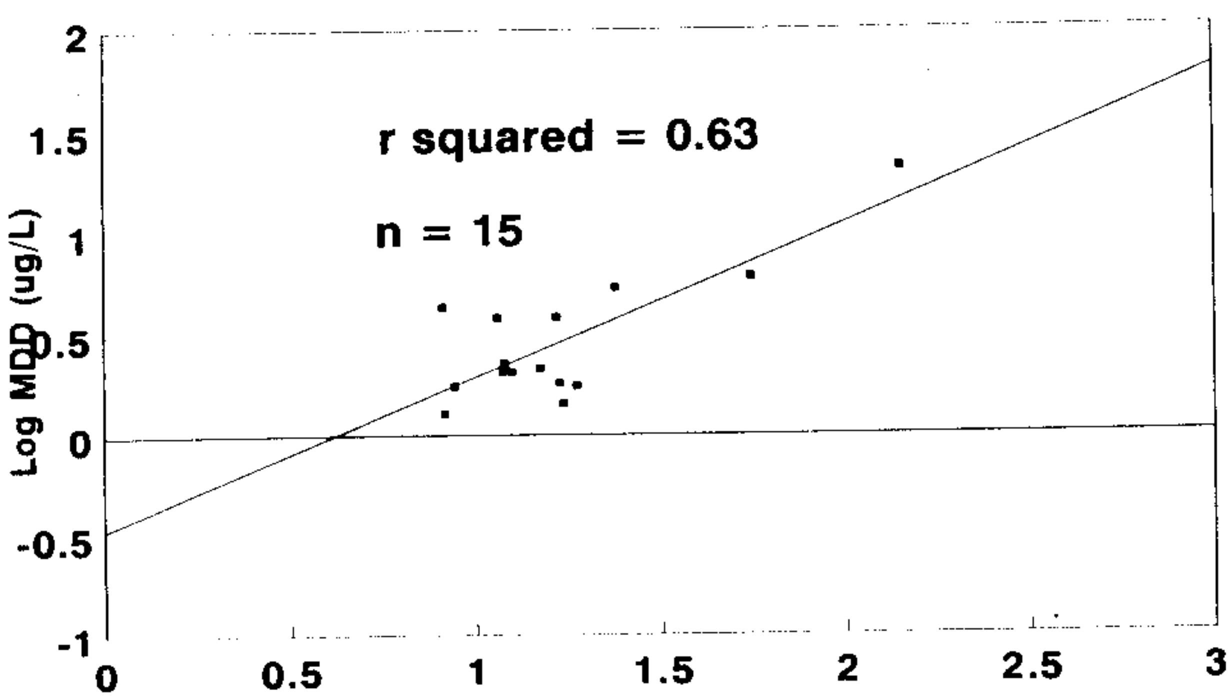
Data from each sample day, when considered as a separate system, provides 15 data points to regress against respective calculated log MDD values. The result of the regression of log MDD versus log chlorophyll a (Figure 1) has a r^2 of 0.63. The influence or weighting of extreme points was determined by elimination of that value, re-regressing the data and comparing the resulting slope to the original for significance using slope comparison (Zar, 1984). The larger number of points in the lower range of chlorophyll a values makes the single high point significantly overweighted in the regression. The regression of non-log transformed MDD values and log chlorophyll a values (Figure 2) improves the r^2 to 0.72, but increases the weight of the higher values even more. A decision was made to use untransformed MDD values in subsequent regressions because the MDD units are the same as the chlorophyll a units and the r^2 was greater. The regression does show application potential by describing 72 percent of variation in MDD with increasing chlorophyll a concentrations. Confidence intervals (95%) were calculated for this regression line (Figure 3) and the resulting window has a width of 11.5 ug/L. The width of the window indicates the degree of sensitivity, or twice the amount of change necessary before a change can be judged to be significant.

Data from July 5 and July 10 were combined as was July 20 and August 5 in Ray Roberts, Kiowa , and Lewisville Lakes. The means of these combinations together with the other systems were regressed with their respective calculated MDD values (Figure 4). The relationship had a r^2 of 0.75 which indicates an improved relationship with the increase in the temporal scale. The 95 percent confidence bands were calculated for this regression (Figure 5). The resulting window had a width of 15.3 ug/L. The increase in r^2 was offset by the increase in window width which means a greater change was necessary for significance. The width of the window increase was due to the decreasing degrees of freedom in the confidence calculation.

Data from all dates combined within each system provides a temporal scale approximating the mean epilimnetic summer chlorophyll a used by OECD in their eutrophication model. This combination resulted in a r^2 of 0.75 (Figure 6) indicating a continued increase in r^2 with the increase in temporal combinations. The six points are not significantly weighted at either end. The 95 percent confidence band window (Figure 7) had a width of 22.3 ug/L. The increase in window width followed the same pattern, as previously described, with an offsetting increase in r^2. Although there are fewer points in this model than the other regressions, the potential of this regression model and subsequent window provides a useful base on which to develop a preliminary chlorophyll a significant difference model.

Figure 1. Regression of log chlorophyll _a_ concentrations and log MDD with each sampling date entered as a separate value.

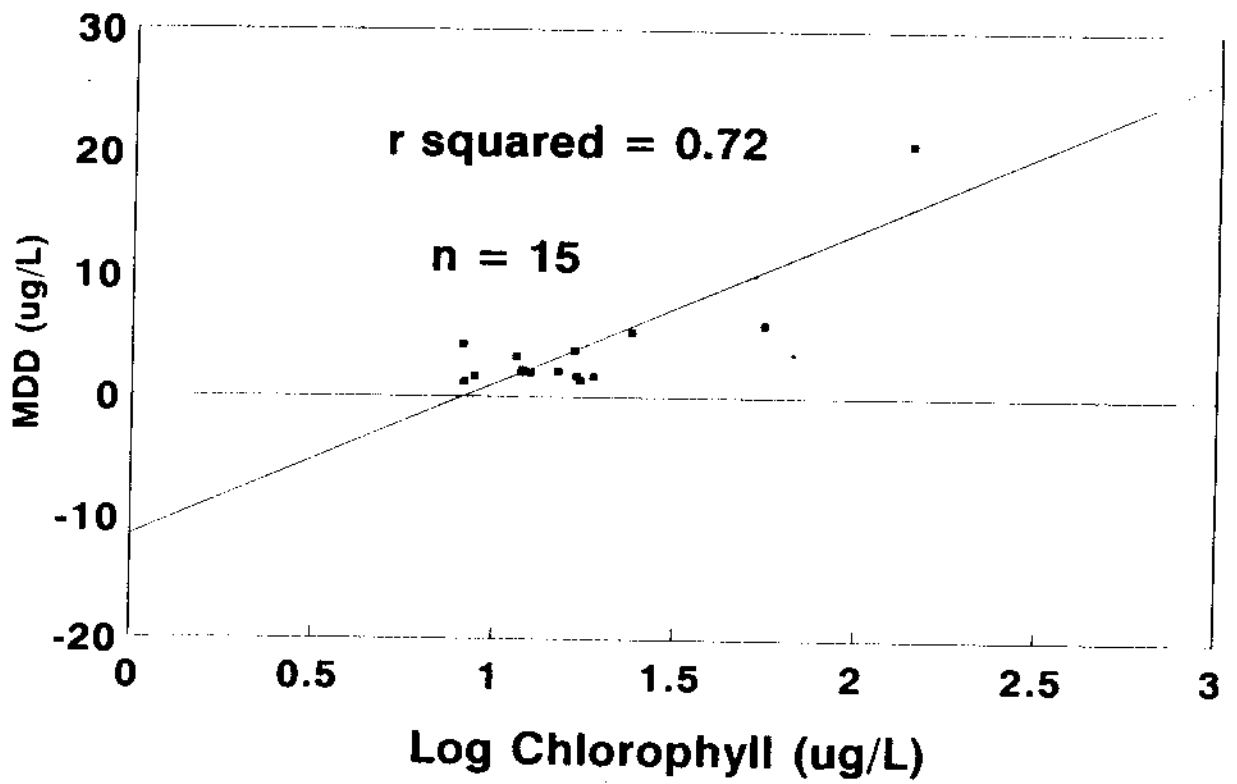
Dates Separate
Log Chl - Log MDD



MDD = -0.481 + 0.763 (log chl.)

Figure 2. Regression of log chlorophyll *a* concentrations and MDD with each sampling date entered as a separate value.

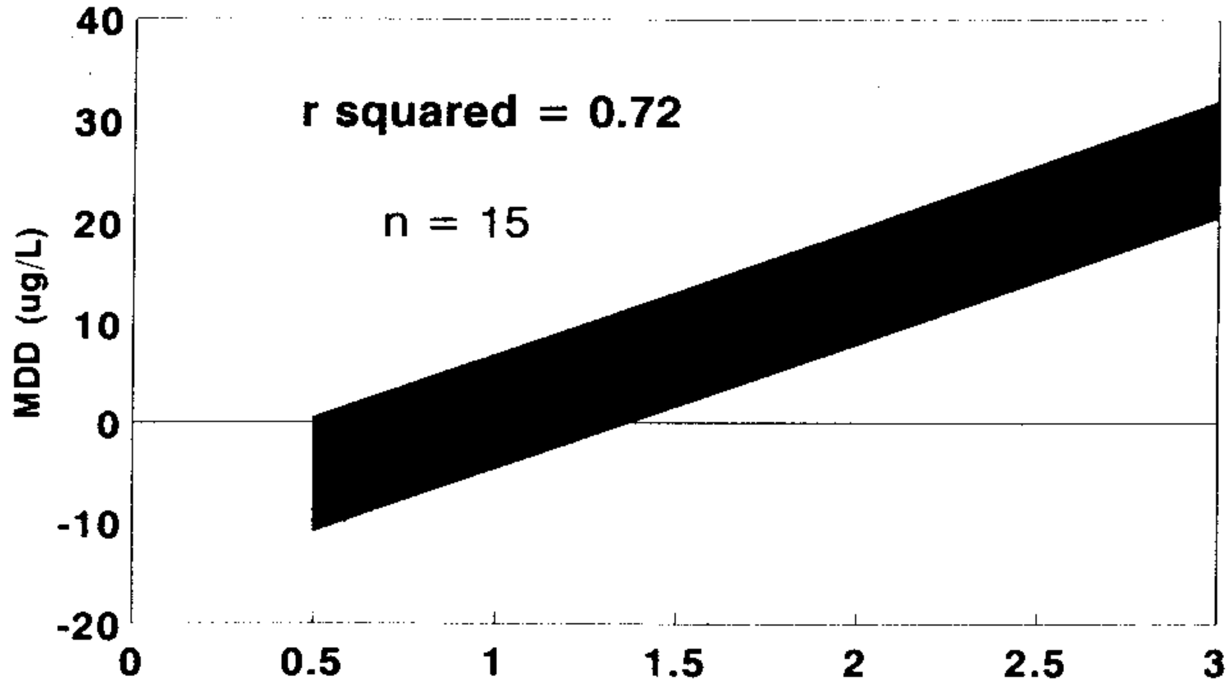
Dates Separate
Log Chl - MDD



MDD = -11.50 + 12.596 (log chl.)

Figure 3. MDD Window derived from 95% mean confidence limits about the regression of log chlorophyll *a* concentrations and MDD.

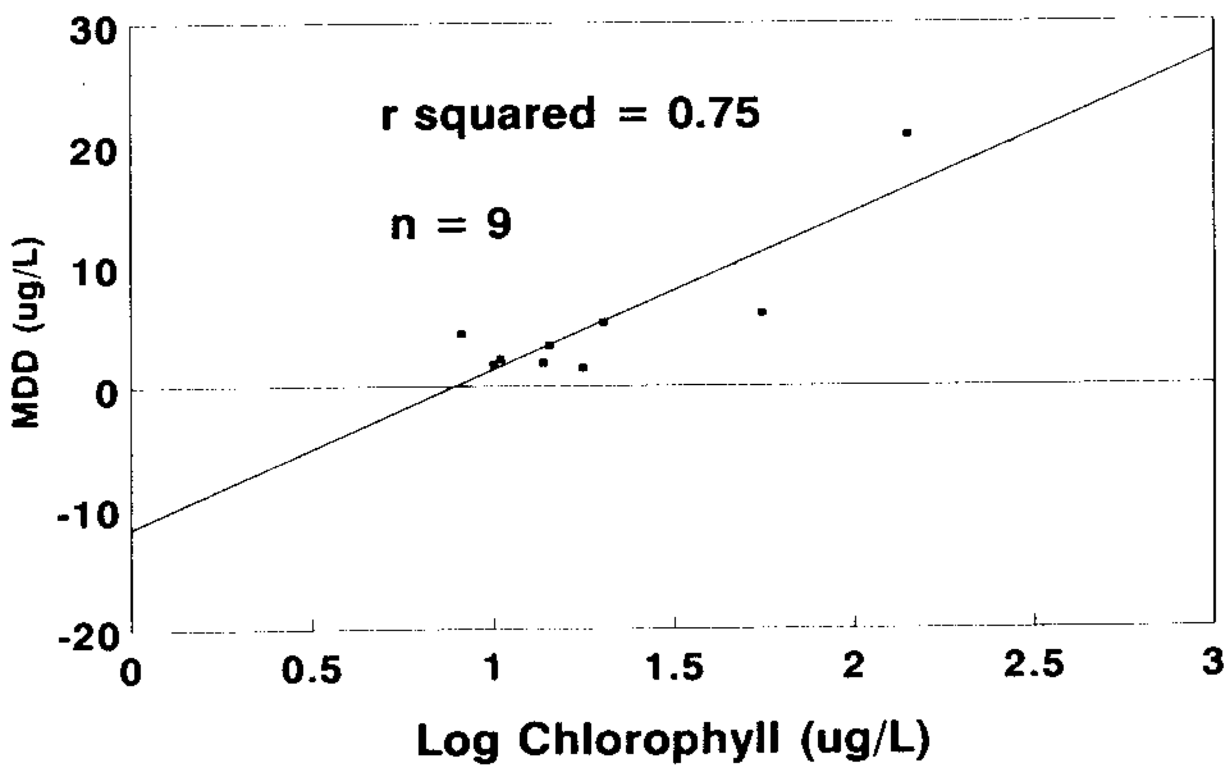
MDD Window
Dates Separate



Model MDD = -5.752 + 12.596 (log chl.)

Figure 4. Regression of log chlorophyll *a* and MDD The July 5 data was combined with the July 10 data and the July 20 data combined with the August 5 data for Ray Roberts, Lewisville, and Kiowa Lakes.

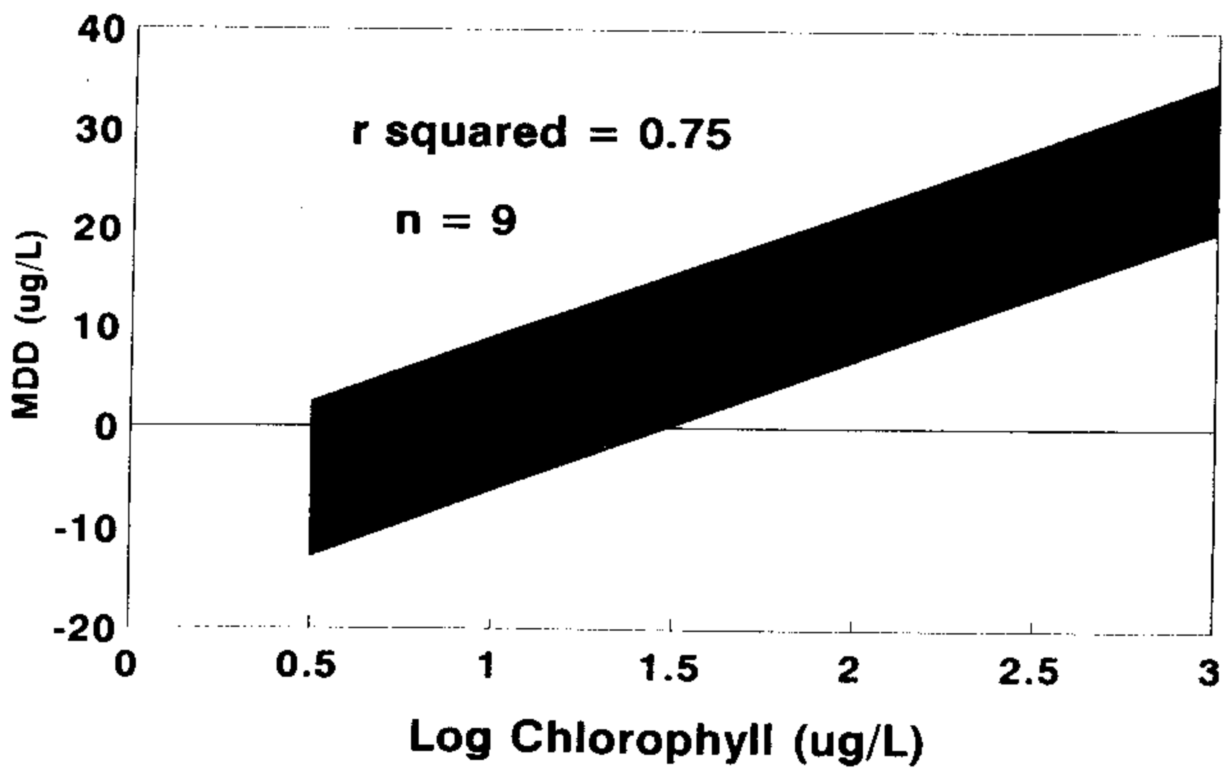
Date Combination
Log Chl. - MDD



MDD = -11.681 + 13056 (log chl.)

Figure 5. MDD Window derived from the 95% mean confidence limits about the two date combination regression of log chlorophyll _a_ and MDD.

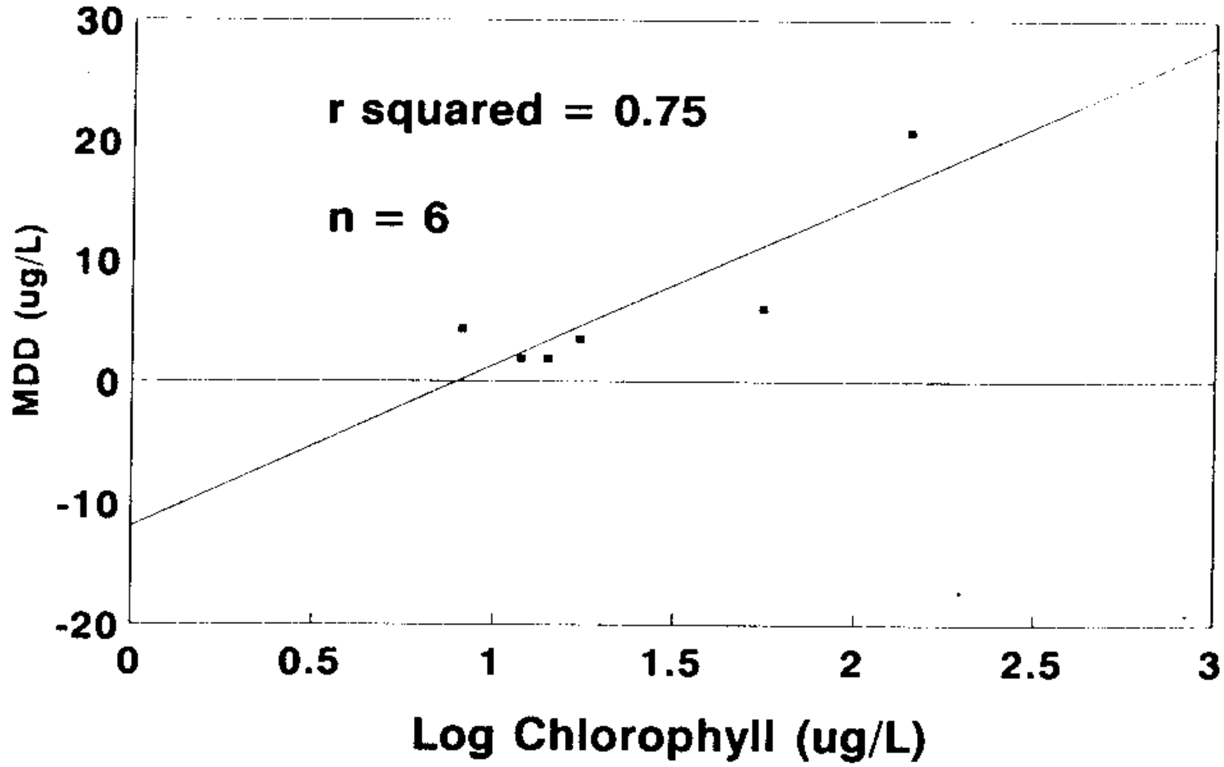
MDD Window
Date Combination



Model MDD = -4.028 + 13.056 (log chl.)

Figure 6. Regression of log chlorophyll *a* concentrations and MDD with all sample dates combined for Ray Roberts, Lewisville, and Kiowa Lakes.

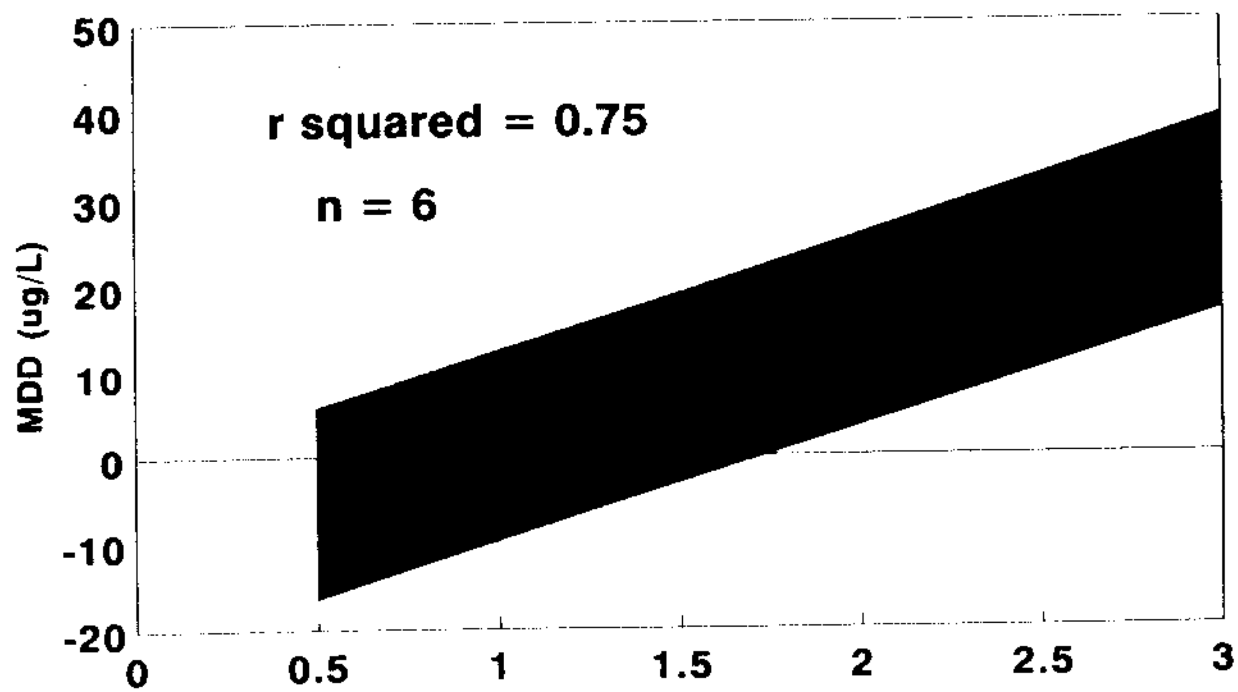
Lake Means
Log Chl. - MDD



MDD = -11.925 + 13.302(log chl.)

Figure 7. MDD Window derived from the 95% mean confidence limits about the regression of all sample date combination.

MDD Window
Lake Means



Model MDD = 0.763 + 13.302(log chl.)

Conclusions

The similarity of all three temporal scale regression results indicates a model that can be potentially applicable at any desired scale. Increases in temporal scale resulted in increased r^2 values, but the differences were not significant. Slopes and elevations for the three regressions were also not significantly different. The difference in window width was the only factor that influenced the potential useful application of the model. These differences stem from the different number of points with which the regressions were calculated and not from differences in the data from which they were derived. Figure 3 appears to best describe a logic relevant to phytoplankton populations. With this model a system with a mean chlorophyll a concentration of 32 ug/L would have to improve by about 11 ug/L (to 21 ug/L) to show significant restoration. The model represented by Figure 7 would require a change of 19 ug/L for the same system.

The results of this effort to develop a variability model showed that the approach has potential as a management tool. The equation described in Figure 3 provides the most logically based model. Investigation of the spatial scaling influences, as well as the temporal influences, are needed to test the reliability of the model. An increased number of data points are needed to test how well the six systems studied represent the relationship of chlorophyll a and its associated variance in all systems. An increased number of data points is also needed to determine the number at which the model sensitivity is too great (the bands too narrow) for practical use.

The social significance of this mathematically described sensitivity is needed to put any change in perspective. How much change can be detected by sight or smell, and is this amount enough to be considered significant? Social significance and relevance is a necessary reference point to statistical significance. Ecological significance is also needed as a reference for any ecological change. Eutrophication indices, such as the community composition indices proposed by Nygaard (1949), have potential as an aid in determining whether a change is indicative of an ecologically relative change in terms of eutrophication. These indices are phytoplankton ratios based on relative abundance of taxa such as centric diatoms, bluegreens, and euglenophyta which are indicative of higher trophic status levels.

The preliminary model presented (Figure 3) exhibits useful potential for standardizing the judgement of statistical significant difference. The benefit in using this model approach is to the many monitoring programs that depend on minimal number of samples (due to budget constraints) to determine significant changes within lentic systems. Given the variation noted in chlorophyll a concentrations based on a sample number of three (number of repetitions at each sample site), a greater magnitude of change would most likely be required for statistical significance than the significance represented by this model. The reverse scenario must also be considered. A lake managers data set for a lake (or lakes) which has low variability may have a lower MDD than that derived from use of the MDD Window. The MDD calculated from such a data set would be more useful for that lake than the MDD derived from the model presented here. The MDD model, as a preliminary tool, can be readapted locally or regionally as the best data sets dictate. Further development of this model is necessary, but the preliminary results indicate a useful management tool.

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CHAPTER VI

DISCUSSION

Historically the trophic status of water bodies had been based on nutrient concentrations (usually total phosphorus), chlorophyll a concentrations, and secchi disk visibility depth. Changes in trophic status, or the potential change in trophic status, are often related to increases or decreases in the allocthonous inputs of nitrogen or phosphorus from changes in land use and management practices. Lake and reservoir managers are continually faced with the questions of what to monitor, how to monitor it, and how much change is necessary to be considered significant. These questions were addressed separately in this study using data from six reservoirs in Texas.

The question of what trophic status indicator to monitor was addressed in Chapter II. Trophic status indicators (TSIs) showed agreement only in the broad trophic classifications of oligotrophic, mesotrophic, and eutrophic. These classifications are not standardized in the demarcation points of the different TSI parameters. The lack of consistency in classification makes the broad categories weak monitoring tools. Chlorophyll a, Secchi disk visibility, and phytoplankton density showed the most consistent agreement. Conversion of TSIs into TSI units developed by Carlson increases the comparability of each index. Phytoplankton community composition indices developed by Nygaard are too broad by themselves to be useful as a monitoring tool. These TSIs are useful, in their present state, as supplemental TSIs because they indicate what could be typical phytoplankton compositions at differing trophic states.

Chlorophyll a concentrations and Secchi disk depth appear (from this study) to be the best trophic status indicators for managers of lakes and reservoirs to monitor because of the level of agreement in their description of trophic status. Chlorophyll a concentrations, of the two, would be a better direct indicator of the turbidity associated with phytoplankton biomass.

The question of how to monitor , or predict, changes in trophic status was addressed in Chapters III and IV. The mean summer epilimnetic chlorophyll a predictions of Ray Roberts Lake using both nutrient loading and OECD eutrophication models (Pillard, 1988) were significantly higher than measured concentrations two years after the impoundment of Ray Roberts Lake. The predicted high and low range values, determined by the high and low nutrient values form the nutrient loading model, were also higher than the measured chlorophyll a concentrations. This study could not determine which of either model's inputs was responsible for the overestimation of mean summer chlorophyll a concentrations or if the overestimation was a function of a lack of stabilization of the reservoir two years after impoundment. The source of the overestimation of epilimnetic chlorophyll a concentrations in Ray Roberts Lake needs to be determined before the use of the OECD eutrophication model is of practical use to the reservoir managers for trophic status prediction and monitoring.

The use of remotely sensed reflectance values for monitoring water quality appeared to be of little practical use to lake managers for the lakes studied. The three lakes studied were not significantly different for any of the clarity parameters measured. The correlation between turbidity and chlorophyll a concentrations indicate that the source of lake turbidity is primarily due to phytoplankton biomass. This supports the conclusion of Chapter II that chlorophyll a is the most practical single trophic endpoint for monitoring and prediction.

Low correlation between water clarity parameter values and band reflectance values appeared to relate directly to the magnitude of difference in chlorophyll a and turbidity values. Previous researchers, such as Lathrop and Lillesand (1989), showed greater than 90 percent correlation of chlorophyll a concentrations with band reflectance. This study did not seem to have a chlorophyll a concentration range wide enough to produce a significant correlation. The use of remotely sensed data appears not to be practical for lake monitoring in those cases where detectable differences are less than the magnitude of difference needed for statistically significant changes in chlorophyll a concentrations.

The development of a preliminary chlorophyll a variability model (Chapter IV) has potential for standardizing lake managers judgement of statistical significant difference in changes in chlorophyll a concentrations. The benefit in using this Minimum Detectable Difference (MDD) model approach for this purpose is that many monitoring programs depend on minimal number of samples to determine significant changes in lakes and reservoirs. The resulting significant difference value of the MDD model may also be used to indicate the amount of sensitivity required for significance in the use of other eutrophication models and other monitoring tools such as remote sensing. Further development of this chlorophyll a MDD model is necessary, but the preliminary results indicate a useful tool for managers of lakes and reservoirs.

The following conclusions correspond to the three hypotheses stated in the Introduction.

1. There is a significant relationship between the minimum detectable difference (MDD) and the

associated mean chlorophyll a concentrations. The regression relationship had an r^2 of 0.75 with the MDD calculated at a significance level of 0.05 (alpha) and a power of 0.1 (beta).

2. There is a difference in the predicted mean summer epilimnetic chlorophyll a concentration and the measured concentrations two years after impoundment of Ray Roberts Lake. Although the expected differences between the Elm Fork arm, the Isle du Bois arm, and the main body of the lake were evident two years after impoundment, the mean predictions and the predicted ranges were consistently higher than the measured concentrations.

3. There was no significant relationship between remotely sensed reflectance values and measured trophic status indicators for Ray Roberts, Lewisville, and Kiowa lakes. The low magnitude of difference between the lakes for all water clarity parameters combined with the high band correlation (greater than 90 percent) resulted in a lack of correlation between band and parameter values.

APPENDIX A

PHYTOPLANKTON TAXONOMIC IDENTIFICATION, ENUMERATION AND WATER QUALITY PARAMETERS FOR STOCKPONDS 2 AND 3

Phytoplankton Enumeration of
Stockponds 2 and Stockpond 3
on August 16, 1991
(Number of organisms per milliliter)

	Rep 1	Rep 2	MEAN
St.P. 2 - 1	116,680	122,880	119,780
St.P. 2 - 2	152,100	134,720	143,410
St.P. 3 - 1	9,960	11,260	10,610
St.P. 3 - 2	9,100	6,820	7,960

Chlorophyll a and Pheophytin a
(in parentheses)
Concentrations of Stockpond 2 and
Stockpond 3 on August 16, 1991
(ug/L)

	Rep 1	Rep 2	MEAN
St.P. 2 - 1	55.14 (0)	57.00 (0)	56.07 (0)
St.P. 2 - 2	57.94 (0)	51.40 (0)	54.67 (0)
St.P. 3 - 1	5.61 (4.21)	10.28 (0.19)	7.94 (2.2)
St.P. 3 - 2	9.35 (7.66)	7.48 (2.34)	8.41 (5.00)

Identification and Enumeration of Phytoplankton in Stockpond 2, sites 1 and 2 (mean number/ml)

	Genus	St.P 2 - 1	St.P 2 - 2
Cyanophyta	Arthrospira	12960	10390
	Raphidiopsis	2780	1330
	Anabaena	1030	570
	Aphanocapsa	11900	9050
	Merismopedia	13740	14020
	Oscillatoria	15230	12230
	Microcystis	56500	91250
Chlorophyta	Chlamydomonas	220	230
	Chlorococcum	80	20
	Cosmarium	280	350
	Crucigenia	880	600
	Scenedesmus	920	1060
	Selenastrum	1600	880
	Staurastrum	60	50
	Tetraedron	40	70
Chrysophyta	Navicula	60	60
	Synedra	10	0
	Tribonema	80	0
Other	Rhodomonas	970	600
	Ceratium	0	10
	Euglena	350	300
	Phacus	40	40
	Lepocinclis	20	40
	Trachelomonas	30	260

Identification and Enumeration of Phytoplankton
in Stockpond 3, sites 1 and 2 (mean number/ml)

	Genus	St.P 3 - 1	St.P 3 - 2
Cyanophyta	Arthrospira	0	0
	Raphidiopsis	0	0
	Anabaena	0	0
	Merismopedia	120	0
	Oscillatoria	130	110
	Microcystis	8500	6850
Chlorophyta	Chlamydomonas	150	70
	Chlorococcum	50	10
	Cosmarium	70	40
	Crucigenia	0	40
	Scenedesmus	300	0
	Kirchneriella	560	510
	Staurastrum	70	40
	Treubaria	10	0
	Tetraedron	0	10
	Schroderia	20	0
	Dictiosphaerium	100	0
	Ankistrodesmus	0	40
	Oocystis	0	10
	Protococcus	0	60
Chrysophyta	Navicula	0	
	Synedra	0	
	Tabellaria	20	
	Chrysococcus	120	
Other	Rhodomonas	90	
	Ceratium	10	
	Euglena	10	
	Phacus	0	
	Trachelomonas	240	

Mean water quality parameters collected in Stockpond 2 and Stockpond 3 on August 16, 1991. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos) secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	Stockpond 2	Stockpond 3
Alkalinity	470	110
Turbidity	17.9	1.9
Dissolved Solids	1364	140
Suspended Solids	24	<1
Chloride	126	6
Sulphate	144	9
Ammonia	<0.05	0.1
Nitrate	<0.1	<0.1
Nitrite	<0.1	<0.1
Total Phosphate	<0.05	<0.05
Orthophosphate	<0.05	<0.05
Secchi Depth	39.6	207.3
pH	9.01 - 9.36	7.90 - 8.98
Temperature	26.5 - 32.8	27.3 - 30.9
Dissolved Oxygen	5.4 - 12.5	3.2 - 10.4
Conductivity	1400 - 1600	220 - 300

APPENDIX B

PHYTOPLANKTON TAXONOMIC IDENTIFICATION, ENUMERATION AND WATER QUALITY PARAMETERS FOR STOCKPOND 1

Water quality parameters for Stockpond 1

Date	Rep	Chloride mg/l	Sulfate mg/l	Total Phosphate mg/l	Ortho Phosphate mg/l
9-28-90	A	551	53.3	0.125	0.063
9-28-90	B	498	56.6	-----	-----
9-28-90	C	523	49.2	-----	-----
9-28-90	AVG	524	53.0	0.125	0.063
9-29-90	A	530	51.3	0.742	0.010
9-29-90	B	519	59.1	-----	-----
9-29-90	C	542	53.8	-----	-----
9-29-90	AVG	530	54.7	0.742	0.010
9-30-90	A	410	52.0	0.725	0.011
9-30-90	B	408	56.6	-----	-----
9-30-90	C	403	56.6	-----	-----
9-30-90	AVG	407	55.1	0.725	0.011

Water quality parameters for Stockpond 1 (cont.)

Date	Rep	Ammonia mg/l	Nitrate (NO_3^-) mg/l	Nitrite (NO_2^-) mg/l
9-28-90	A	0.023	0.812	<0.007
9-28-90	B	0.020	0.466	<0.007
9-28-90	C	0.019	0.797	<0.007
9-28-90	AVG	0.021	0.692	<0.007
9-29-90	A	0.59	2.54	<0.007
9-29-90	B	0.61	2.17	<0.007
9-29-90	C	0.62	2.19	<0.007
9-29-90	AVG	0.61	2.30	<0.007
9-30-90	A	0.031	1.22	<0.007
9-30-90	B	0.024	0.872	<0.007
9-30-90	C	0.022	0.720	<0.007
9-30-90	AVG	0.026	0.937	<0.007

Water quality parameters for Stockpond 1 (cont.)

Date	Rep	TVS mg/l	TVSS mg/l	Total Solids mg/l	TSS mg/l	Turbidity NTU
9-28-90	A	176	96	846	114	80
9-28-90	B	171	97	856	115	78
9-28-90	C	156	100	843	119	85
9-28-90	AVG	168	98	848	116	81
9-29-90	A	148	85	844	100	60
9-29-90	B	158	85	843	101	59
9-29-90	C	154	87	866	102	59
9-29-90	AVG	153	86	851	101	59
9-30-90	A	151	81	842	96	58
9-30-90	B	169	81	837	95	59
9-30-90	C	177	82	848	97	57
9-30-90	AVG	166	81	842	96	58

Water quality parameters for Stockpond 1 (cont.)

Date	Rep	Alkalinity mg/l	Hardness mg/l	Chlorophyll mg/m^3	Pheophyton mg/m^3
9-28-90	A	200	144	130.8	0.0
9-28-90	B	205	152	141.1	0.0
9-28-90	C	205	136	137.4	0.0
9-28-90	AVG	203	144	136.4	0.0
9-29-90	A	215	148	159.8	0.0
9-29-90	B	210	144	169.1	0.0
9-29-90	C	215	148	164.5	0.0
9-29-90	AVG	213	147	164.5	0.0
9-30-90	A	210	164	117.7	1.3
9-30-90	B	210	156	121.5	0.0
9-30-90	C	210	160	127.1	0.0
9-30-90	AVG	210	160	122.1	0.4

Water quality parameters for Stockpond 1 (cont.)

Date	Time	Depth	Temp. °C	Dissolved Oxygen mg/l	Conductivity umoles/cm^2
9-28-90	4:30 p.m.	Surface	32	10.8	1600
9-29-90	9:00 a.m.	Surface	26	1.6	1500
9-29-90	9:00 a.m.	Bottom	26	0.8	1500
9-29-90	4:00 p.m.	Surface	32	10.0	1500
9-29-90	4:00 p.m.	Bottom	32	13.0	1500
9-30-90	7:30 a.m.	Surface	26	0.2	1500
9-30-90	7:30 a.m.	Bottom	26	0.2	1500

Identification and Enumeration of Phytoplankton in Stockpond 1 (number ml^{-1})

	Genus-Species	9-28-90	9-29-90	9-30-90
CYNANOPHYTA	*Anabaenopsis circularis*	1631	3805	1087
	Arthrospira massartii	10328	13046	7339
	Lyngbya limnetica	4969	37542	11042
	Merismopedia tenuissima	59623	172247	72874
	Microcystis incerta	99372	44166	30916
	Oscillatoria amphibia	9937	-----	6625
	Oscillatoria angusta	39749	50791	83915
	Oscillatoria tenuis	422331	423994	388661
	Raphidiopsis mediterranea	142433	161206	163414

CHLOROPHYTA	*Actinastrum hantzschii*	1631	------	4349
	Ankistrodesmus falcatus	1359	1631	2446
	Ankistrodesmus spiralis	815	544	272
	Chlamydomonas	1656	------	4417
	Dictyosphaerium pulchellum	1087	1087	1087
	Golenkenia radiata	272	272	272
	Monoraphidium circinale	1656	2208	2208
	Scenedesmus acuminatus	3262	3533	7339
	Scenedesmus quadricauda	------	1087	-----
	Selenastrum minutum	6625	13250	8833
	Coccoid chlorophyta	28155	52999	39749
	Coccoid chlorophyta	3312	4417	2208
CRYPTOPHYTA	*Cryptomoms erosa*	5980	2990	3805
	Rhodomonas minuta	------	------	2208
EUGLENOPHYTA	*Euglena spp.*	------	------	272
CHRYSOPHYTA	*Chrysoflagellate*	6625	26500	44166
	Chrysoflagellate	------	-----	11042
	Coccoid chrysophyta	1656	11042	2208

APPENDIX C

WATER CLARITY PARAMETERS FOR RAY ROBERTS, LEWISVILLE, AND KIOWA LAKES ON JULY 5, JULY 10, JULY 20, AND AUGUST 5

Mean water clarity parameter values of Ray Roberts Lake on July 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	7.94	1.22	2	16	171
2	9.35	3.08	3	28	169
3	13.08	1.08	3	21	174
4	11.68	0.75	3	18	177
5	14.49	1.87	3	18	173
6	13.08	3.27	4	20	167
7	25.70	2.43	4	19	170
8	13.55	2.15	3	18	182
9	11.68	3.04	3	18	185
10	9.81	3.27	6	17	163
11	23.36	6.07	5	17	189
12	15.89	5.05	5	20	178
13	11.68	1.73	16	22	164
14	15.89	2.10	4	*	*
15	7.01	1.82	4	14	167
16	9.35	2.10	3	21	161
17	9.35	1.78	5	10	189
18	9.35	1.45	5	13	178
19	15.89	2.10	6	13	169
20	18.69	0.33	5	14	173
21	11.68	1.51	5	*	*
22	7.01	0.75	4	*	*
23	9.35	1.10	4	*	*
24	10.28	1.17	3	15	175

* = no data

Mean water clarity parameter values of Ray Roberts Lake on July 10, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	12.15	0.75	3	7	181
2	13.55	0.84	2	7	169
3	17.29	0.09	4	22	97
4	15.42	3.41	8	6	174
5	13.55	3.13	5	5	173
6	18.69	2.43	5	18	91
7	15.89	0.75	4	2	167
8	16.35	0.80	4	20	110
9	18.22	0	3	3	217
10	29.44	0	5	4	126
11	13.08	3.93	4	2	181
12	12.15	0.94	3	0	180
13	15.42	0.42	5	2	179
14	20.56	0	5	2	179
15	10.28	0.92	8	6	183
16	15.89	0	8	26	117
17	8.88	2.24	7	1	161
18	12.62	0.89	12	0	156
19	11.21	0.84	4	1	160
20	14.95	0	6	1	182
21	11.21	0.61	7	2	167
22	11.68	1.40	4	2	164
23	12.62	0.14	4	13	137
24	16.82	1.64	3	1	162

Mean water clarity parameter values of Ray Roberts Lake on July 20, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	7.01	3.74	6	6	175
2	6.08	5.70	6	6	179
3	4.67	6.45	7	7	176
4	9.81	6.22	9	2	180
5	9.81	2.62	7	7	175
6	7.48	1.03	5	6	178
7	7.48	3.32	4	6	229
8	4.67	4.16	5	4	186
9	7.01	4.77	5	4	197
10	14.02	0	6	11	164
11	9.81	0	7	14	166
12	10.28	0.23	8	7	178
13	12.62	0.94	8	5	201
14	8.41	1.73	7	6	201
15	7.48	2.66	5	8	130
16	7.48	2.66	6	4	161
17	5.14	3.04	7	4	164
18	5.14	1.12	8	7	149
19	7.48	0.19	11	10	160
20	11.21	0	10	14	184
21	4.68	1.73	7	14	126
22	9.35	1.82	7	18	181
23	7.48	0.70	5	11	180
24	14.02	0	6	11	174

Mean water clarity parameter values of Ray Roberts Lake on August 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	7.94	1.22	2	4	171
2	12.15	1.59	1	4	163
3	7.94	4.16	1	11	164
4	8.41	0.42	3	9	175
5	4.21	5.28	2	5	170
6	8.41	2.71	6	5	174
7	8.88	1.92	3	4	167
8	7.01	0.84	4	4	173
9	10.75	2.34	5	2	174
10	9.35	4.11	4	7	189
11	12.15	3.88	3	5	186
12	10.28	1.82	3	6	166
13	10.75	3.97	11	10	199
14	6.54	3.93	4	2	170
15	15.42	4.21	5	8	178
16	18.69	2.24	5	6	166
17	14.95	4.35	6	6	178
18	19.63	2.62	4	6	167
19	17.29	3.65	9	9	178
20	21.96	3.55	9	17	207
21	18.22	3.36	5	11	181
22	16.35	4.58	6	13	173
23	7.48	5.28	6	11	184
24	9.81	3.93	6	9	184

Mean water clarity parameter values of Lewisville Lake on July 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	6.08	2.76	3	3	200
2	14.49	0.51	4	6	178
3	7.48	2.34	3	3	194
4	10.75	0.89	4	0	195
5	13.55	2.94	4	5	184
6	16.82	2.48	4	3	184
7	11.21	7.94	4	2	176

Mean water clarity parameter values of Lewisville Lake on July 10, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	19.16	3.76	7	*	*
2	15.89	2.14	8	*	*
3	15.42	4.83	8	*	*
4	17.76	3.33	11	*	*
5	16.82	3.79	10	*	*
6	18.69	2.02	12	*	*
7	16.82	3.80	12	*	*

* = no data

Mean water clarity parameter values of Lewisville Lake on July 20, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	15.42	0.28	9	3	192
2	17.76	2.85	10	4	191
3	11.21	3.83	7	3	164
4	21.96	0.28	10	4	170
5	15.42	3.22	10	2	165
6	18.69	1.59	10	3	177
7	15.42	0	10	4	176

Mean water clarity parameter values of Lewisville Lake on August 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	30.84	3.83	18	37	218
2	35.04	2.90	16	34	214
3	22.90	2.94	7	21	178
4	21.03	2.20	6	20	170
5	17.76	1.45	5	21	204
6	17.76	1.87	11	32	222
7	21.96	0.37	4	18	198

Mean water clarity parameter values of Kiowa Lake on July 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	9.35	0.05	11	6	200
2	7.94	0	10	8	178
3	6.08	2.24	10	7	194
4	8.41	0.23	11	12	195
5	7.01	0.47	8	7	184
6	10.28	0.89	10	6	184
7	11.21	0.23	10	4	176

Mean water clarity parameter values of Kiowa Lake on July 10, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	8.88	2.90	11	7	150
2	11.21	2.20	12	3	189
3	13.55	0.98	7	6	198
4	16.82	0	7	4	179
5	12.15	1.12	9	3	185
6	12.62	0.47	7	1	180
7	10.28	2.80	7	0	180

Mean water clarity parameter values of Kiowa Lake on July 20, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	14.95	0.03	12	19	180
2	16.35	0	11	15	182
3	15.89	3.08	10	16	180
4	19.63	1.96	10	13	185
5	15.42	1.64	12	19	181
6	17.29	2.99	10	13	182
7	18.22	1.73	10	14	187

Mean water clarity parameter values of Kiowa Lake on August 5, 1989. Chlorophyll and pheophytin in ug/L, TSS and TDS in mg/L and turbidity in NTU.

Site	Chl. a	Pheo. a	Turb.	TSS	TDS
1	19.63	0.98	10	34	194
2	20.09	0	10	16	197
3	18.69	0	7	14	198
4	20.56	0.65	8	17	195
5	17.29	0.19	13	21	198
6	18.22	1.26	6	13	194
7	16.35	3.60	6	11	179

Mean band reflectance values of Ray Roberts Lake on July 20, 1989.

Site	Band 1	Band 2	Band 3
1	26.76	60.84	41.06
2	30.07	66.75	45.91
3	32.82	71.30	50.07
4	28.82	66.13	45.32
5	27.21	63.67	43.20
6	31.73	69.58	47.96
7	27.81	64.53	43.84
8	31.03	67.23	46.65
9	28.31	65.02	44.06
10	27.39	60.64	41.53
11	29.84	66.79	46.23
12	31.98	70.51	49.11
13	29.03	67.54	45.31
14	30.95	69.84	47.90
15	30.54	69.95	48.92
16	29.00	66.79	45.88
17	32.15	68.15	46.48
18	30.99	70.49	48.31
19	30.69	70.79	48.16
20	29.03	68.64	45.92
21	27.69	64.17	42.71
22	27.88	66.09	44.09
23	29.87	69.42	48.57
24	24.83	58.50	36.67

Mean band reflectance values of Lewisville Lake on July 20, 1989.

Site	Band 1	Band 2	Band 3
1	36.81	75.29	54.16
2	34.60	79.38	57.05
3	37.05	78.05	55.17
4	34.05	77.26	54.63
5	26.44	68.06	45.15
6	34.08	78.83	55.37
7	35.99	81.61	58.12

Mean band reflectance values of Kiowa Lake on July 20, 1989.

Site	Band 1	Band 2	Band 3
1	37.37	78.34	56.36
2	35.38	75.22	53.22
3	33.48	72.78	50.93
4	35.47	75.86	53.11
5	35.58	76.44	54.81
6	34.39	73.28	51.33
7	31.84	70.53	48.50

APPENDIX D

PHYTOPLANKTON TAXONOMIC IDENTIFICATION AND ENUMERATION

FOR RAY ROBERTS LAKE QUARTERLY SAMPLING

AND WATER QUALITY PARAMETERS

COLLECTED MONTHLY

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos), secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
June 1989									
Alkalinity	98	100	98	88	108	95	145	82	98
Hardness	107	89	109	101	109	113	173	120	156
Turbidity	12	16	10	4	5	7	64	72	107
Dissolved Solid	179	172-	173--	133	189	194	245	245	236
Suspended Solid	19	20	15	5	13	9	112	95	137
Chloride	27	29	30	33	33	32	34	83	29
Sulfate	20	23	21	23	21	21	43	48	95
Ammonia	0.05	0.04	0.05	0.07	0.06	0.02	0.04	0.04	0.03
Nitrate	0.85	1.21	1.18	0.53	0.84	1.03	1.77	0.05	1.62
Total Phosphate	*	*	*	*	*	*	*	*	*
Orthophosphate	0.05	0.06	0.05	0.00	0.06	0.03	0.02	0.05	0.09
Chlorophyl	10	9	9	7	7	9	7	5	8
Pheophytin	4	1	0	1	2	1	4	5	4
Secchi Depth	58	66	87	106	109	104	NA	NA	NA
1 % Light	*	*	*	*	*	*	NA	NA	NA
pH	7.8	7.9	7.8	7.8	7.8	7.7	7.8	7.9	7.8
Temperature	28	25	25	24	24	25	19	19	22
Dissolved Oxygen	9.2	6.5	6.9	6.4	6.8	6.6	8.3	8.1	7.9
Conductivity	270	275	290	260	280	285	520	550	310

* = no data - = # of reps

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
July 1989									
Alkalinity	107	102	102	100	100	98	238	145	133
Hardness	111	119	108	103	105	125	229	225	147
Turbidity	4	5	4	4	3	3	10	11	28
Dissolved Solids	174	181	178	175	197	166	220	230	187
Suspended Solids	11	18	7	6	4	4	24	12	17
Chloride	23	26	30	28	29	31	108	289	80
Sulfate	11	13	9	12	10	17	54	18	50
Ammonia	0.07	0.05	0.06	0.05	0.06	0.06	0.08	0.06	0.19
Nitrate	0.39	0.36	0.37	0.45	0.38	0.33	5.94	0.64	0.99
Total Phosphate	0.11	0.03	0.11	0.13	0.02	0.07	0.76	0.50	0.19
Orthophosphate	0.03	0.03	0.00	0.00	0.00	0.00	0.73	0.01	0.07
Chlorophyl	14	9	10	7	7	7	10	5	12
Pheophytin	0	2	0	4	5	3	4	4	6
Secchi Depth	132	141	160	152	156	160	NA	NA	NA
1 % Light	4.1	4.5	4.5	4.5	4.5	4.4	NA	NA	NA
pH	8.0	7.5	7.2	7.5	7.5	7.4	7.8	7.8	7.6
Temperature	32	31	29	30	28	30	26	24	27
Dissolved Oxygen	7.2	6.4	7.4	7.2	7.4	7.4	7.9	8.1	7.5
Conductivity	310	320	295	280	300	300	685	630	390

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos), secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
August 1989									
Alkalinity	115	115	108	107	107	109	242	*	138
Hardness	125	119	109	109	112	111	257	*	144
Turbidity	6	6	3	2	3	5	10	*	27
Dissolved Solids	184	173	166	171	167	174	*	*	*
Suspended Solids	9	13	7	4	4	2	46	*	46
Chloride	29	33	34	34	27	25	112	*	62
Sulphate	10	10	10	10	10	10	66	*	75
Ammonia	0.05	0.04	0.03	0.03	0.03	0.03	0.04	*	0.23
Nitrate	0.25	0.21	0.19	0.18	0.18	0.18	0.42	*	0.73
Total Phosphate	0.00	0.00	0.00	0.00	0.00	0.00	0.93	*	0.20
Orthophosphate	0.00	0.00	0.00	0.00	0.00	0.00	1.13	*	0.10
Chlorophyl	10	16	10	8	11	9	11	*	52
Pheophytin	4	5	2	1	2	2	5	*	9
Secchi Depth	100	120	120	150	170	135	NA	NA	NA
1 % Light	3.0	3.5	3.5	4.0	4.0	3.5	NA	NA	NA
pH	7.9	8.1	7.8	7.6	7.6	7.1	7.7	*	7.7
Temperature	29	28	28	30	30	29	24	*	25
Dissolved Oxygen	7.2	8.1	6.6	6.3	7.6	8.4	8.3	*	8.2
Conductivity	335	315	310	300	300	300	900	*	420
September 1989									
Alkalinity	118	109	105	92	108	102	182	*	135
Hardness	122	119	116	113	108	109	180	*	152
Turbidity	2	2	1	2	2	2	18	*	25
Dissolved Solids	199	201	201	181	211	196	367	*	312
Suspended Solids	2	3	7	9	5	7	25	*	49
Chloride	39	41	41	39	44	43	73	*	86
Sulphate	10	10	8	10	8	9	33	*	0.35
Ammonia	0.06	0.06	0.10	0.09	0.06	0.05	0.06	*	0.13
Nitrate	0.55	0.43	0.41	0.35	0.41	0.35	9.46	*	1.65
Total Phosphate	0.04	0.03	0.01	0.02	0.04	0.06	0.83	*	0.25
Orthophosphate	0.00	0.01	0.00	0.00	0.01	0.00	0.70	*	0.07
Chlorophyl	13	10	6	11	20	9	13	*	50
Pheophytin	7	6	6	6	9	3	6	*	6
Secchi Depth	150	150	185	168	150	162	NA	NA	NA
1 % Light	4.8	5.0	5.2	5.4	5.2	5.2	NA	NA	NA
pH	8.3	7.7	6.9	6.9	7.9	7.3	7.7	*	7.9
Temperature	25	27	27	26	26	26	24	*	25
Dissolved Oxygen	9.5	8.6	6.0	7.2	10.0	8.0	8.5	*	9.0
Conductivity	255	290	295	260	290	295	720	*	340

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos), secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
October 1989									
Alkalinity	138	112	112	107	110	112	273	*	125
Hardness	124	125	116	77	117	113	175	*	147
Turbidity	5	5	6	4	5	6	12	*	24
Dissolved Solids	175	178	157	118	166	179	550	*	258
Suspended Solids	15	35	21	6	19	9	18	*	34
Chloride	32	36	36	36	34	33	93	*	60
Sulphate	9	10	9	9	9	9	65	*	41
Ammonia	0.07	0.09	0.12	0.09	0.09	0.08	0.03	*	0.02
Nitrate	1.29	1.39	1.41	1.25	1.43	1.28	16.85	*	1.46
Nitrite	0.01	0.00	0.00	0.00	0.00	0.00	*	*	*
Total Phosphate	0.04	0.02	0.02	0.03	0.04	0.03	1.29	*	0.15
Orthophosphate	0.00	0.00	0.00	0.00	0.00	0.00	1.16	*	0.04
Chlorophyl	10	8	9	12	10	9	11	*	56
Pheophytin	9	6	7	7	6	7	5	*	2
Secchi Depth	120	125	130	175	145	125	NA	NA	NA
1 % Light	3.4	3.6	3.6	4.7	4.2	3.8	NA	NA	NA
pH	7.5	7.3	7.3	7.3	7.2	7.3	7.7	*	7.5
Temperature	20	20	20	20	19	20	18	*	19
Dissolved Oxygen	9.0	9.4	8.8	7.6	8.4	9.0	9.0	*	8.9
Conductivity	260	260	260	250	260	260	685	*	325
November 1989									
Alkalinity	123	115	110	113	105	108	292	*	122
Hardness	120	117	117	108	117	108	196	*	153
Turbidity	10	10	10	7	6	8	9	*	28
Dissolved Solids	74	82	98	112	114	101	473	*	195
Suspended Solids	5	8	19	14	15	12	7	*	29
Chloride	40	36	37	37	35	36	107	*	56
Sulphate	12	10	11	11	11	11	62	*	47
Ammonia	0.11	0.11	0.10	0.15	0.09	0.08	0.11	*	0.12
Nitrate	1.17	1.44	1.59	1.28	1.54	1.72	10.50	*	2.94
Nitrite	0.00	0.00	0.00	0.00	0.00	0.00	*	*	*
Total Phosphate	0.05	0.36	0.11	0.04	0.02	0.05	1.19	*	0.03
Orthophosphate	0.00	0.00	0.00	0.00	0.00	0.00	1.06	*	0.02
Chlorophyl	10	8	9	9	8	8	9	*	22
Pheophytin	5	4	4	4	4	5	4	*	9
Secchi Depth	98	100	106	145	157	116	NA	NA	NA
1 % Light	3.0	3.0	3.2	4.6	4.8	3.5	NA	NA	NA
pH	7.0	7.5	7.5	7.5	7.5	7.6	7.8	*	7.9
Temperature	15	15	15	16	15	15	11	*	14
Dissolved Oxygen	9.8	9.8	8.6	8.8	10.0	9.0	8.6	*	9.7
Conductivity	270	260	260	310	280	260	600	*	310

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos) secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
December 1989									
Alkalinity	87	97	85	97	70	100	258	*	83
Hardness	115	115	111	121	104	105	220	*	128
Turbidity	4	4	5	3	3	3	5	*	8
Dissolved Solids	149	148	134	136	139	143	524	*	176
Suspended Solids	9	3	13	12	3	3	13	*	23
Chloride	30	30	28	28	32	35	131	*	41
Sulphate	9	10	10	10	10	10	77	*	27
Ammonia	0.04	0.04	0.02	0.02	0.02	0.02	0.02	*	0.02
Nitrate	1.09	1.27	1.34	1.26	1.38	1.41	10.57	*	1.48
Nitrite	0.03	0.03	0.02	0.04	0.03	0.02	0.04	*	0.03
Total Phosphate	0.06	0.23	0.03	0.02	0.03	0.04	1.20	*	0.02
Orthophosphate	0.01	0.00	0.00	0.01	0.02	0.00	1.28	*	0.01
Chlorophyl	6	5	5	5	4	5	4	*	6
Pheophytin	5	6	7	6	7	6	6	*	5
Secchi Depth	138	154	170	187	189	180	NA	NA	NA
1 % Light	4.3	4.7	4.7	5.0	5.0	4.8	NA	NA	NA
pH	7.8	7.7	7.6	7.5	7.5	7.0	7.7	*	8.5
Temperature	14	14	14	14	14	14	12	*	15
Dissolved Oxygen	11.4	12.4	11.4	10.4	11.2	11.4	13.0	*	10.8
Conductivity	235	235	235	230	230	235	800	*	260
January 1990									
Alkalinity	117	123	107	110	113	112	252	103	122
Hardness	123	127	127	123	123	125	187	176	172
Turbidity	4	2	2	2	3	2	18	111	31
Dissolved Solid	181	201	165	173	111	188	499	429	306
Suspended Solid	34	35	33	6	4	5	22	161	59
Chloride	33	34	35	33	33	32	101	202	50
Sulphate	11	10	10	10	11	11	64	60	73
Ammonia	0.02	0.02	0.02	0.02	0.02	0.02	0.03	0.02	0.02
Nitrate	2.20	2.18	2.85	2.17	2.47	2.61	14.53	2.69	4.12
Nitrite	0.01	0.01	0.01	0.01	0.01	0.00	0.06	0.01	0.01
Total Phosphate	0.11	0.50	0.33	0.02	0.01	0.05	1.46	0.18	0.05
Orthophosphate	0.00	0.00	0.01	0.00	0.00	0.00	1.44	0.00	0.01
Chlorophyl	17	3	7	14	13	8	13	7	10
Pheophytin	4	10	5	6	6	5	10	4	5
Secchi Depth	134	179	183	193	193	184	NA	NA	NA
1 % Light	4.1	5.1	5.3	5.5	5.5	5.4	NA	NA	NA
pH	7.7	8.0	7.6	8.4	8.1	7.8	7.4	7.2	8.1
Temperature	11	9	10	10	8	8	17	17	12
Dissolved Oxygen	11	11.5	11.2	11.0	11.8	11.8	10.5	9.3	11
Conductivity	220	220	215	200	210	210	680	510	340

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos) secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
February 1990									
Alkalinity	123	110	112	107	110	110	207	135	115
Hardness	132	120	124	117	120	121	231	229	125
Turbidity	5	4	3	3	3	3	16	17	9
Dissolved Solid	218	199	184	185	190	213	215	173	133
Suspended Solid	19	13	7	8	7	5	85	100	17
Chloride	32	29	32	30	30	30	105	196	34
Sulphate	13	12	11	12	11	12	50	27	15
Ammonia	0.02	0.01	0.01	0.04	0.02	0.01	0.01	0.01	0.01
Nitrate	1.61	1.46	1.50	1.31	1.39	1.34	5.80	1.58	1.47
Nitrite	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total Phosphate	0.13	0.27	0.06	0.14	0.12	0.61	1.18	0.07	0.06
Orthophosphate	0.01	0.01	0.00	0.00	0.00	0.00	0.88	0.00	0.01
Chlorophyl	14	14	5	10	5	1	22	4	5
Pheophytin	11	7	8	7	0	7	11	8	8
Secchi Depth	104	150	150	156	156	150	NA	NA	NA
1 % Light	3.0	5.0	5.0	5.2	5.3	5.1	NA	NA	NA
pH	7.8	7.7	8.1	8.2	8.3	8.2	7.7	7.5	7.8
Temperature	11	11	10	11	9	10	10	9	10
Dissolved Oxygen	11	11.0	11.0	10.2	10.6	11.4	10.0	9.8	9.8
Conductivity	255	240	220	220	230	230	575	575	270
March 1990									
Alkalinity	112	118	115	110	132	113	175	158	140
Hardness	117	117	121	119	119	119	201	235	163
Turbidity	16	14	6	4	6	6	19	24	54
Dissolved Solid	201	188	184	161	183	191	323	525	293
Suspended Solid	45	33	18	15	16	21	41	28	103
Chloride	53	54	52	50	53	43	81	339	90
Sulphate	12	12	10	10	12	12	27	32	43
Ammonia	0.19	0.32	0.10	0.10	0.12	0.19	0.08	0.11	0.14
Nitrate	1.75	1.71	1.66	1.60	2.07	1.99	4.78	4.08	4.23
Nitrite	0.02	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Total Phosphate	0.19	0.11	0.19	0.14	0.13	0.15	0.54	0.92	0.65
Orthophosphate	0.01	0.02	0.00	0.00	0.00	0.00	0.10	0.02	0.00
Chlorophyl	12	7	5	10	5	1	5	1	2
Pheophytin	6	7	9	7	6	9	0	6	11
Secchi Depth	65	84	130	170	155	130	NA	NA	NA
1 % Light	2.0	2.5	4.7	5.0	5.0	5.0	NA	NA	NA
pH	8.1	8.1	8.0	8.1	8.1	8.1	8.2	7.8	8.0
Temperature	15	15	15	15	14	15	13	13	15
Dissolved Oxygen	9.4	9.6	10.2	10.8	10.2	10.2	10.8	10.6	10.2
Conductivity	255	250	250	240	240	250	290	550	345

Water quality parameters collected during monthly surveys at each lake site 1989 to 1990. All values are means expressed in mg/L except turbidity (NTU), pH (pH units), temperature (degrees C), 1% light (meters), conductivity (umhos) secchi depth (cm.), and chlorophyl and pheophytin (ug/L).

	EF1	EF2	EF3	ID1	ID2	ID3	TR	TC	BTR
April 1990									
Alkalinity	113	110	115	113	110	118	147	72	115
Hardness	117	117	116	117	113	115	175	97	121
Turbidity	12	7	4	4	3	6	50	76	6
Dissolved Solid	163	181	153	134	107	124	215	173	133
Suspended Solid	17	11	11	5	12	13	85	67	17
Chloride	51	52	53	53	51	52	82	104	54
Sulphate	15	12	12	14	15	12	9	12	12
Ammonia	0.02	0.01	0.02	0.01	0.02	0.01	0.01	0.02	0.01
Nitrate	0.24	0.20	0.19	0.16	0.16	0.18	0.46	0.11	0.20
Nitrite	0.05	0.02	0.01	0.03	0.03	0.00	0.04	0.02	0.02
Total Phosphate	*	*	*	*	*	*	*	*	*
Orthophosphate	*	*	*	*	*	*	*	*	*
Chlorophyl	12	6	3	7	6	2	8	7	3
Pheophytin	10	6	9	7	7	7	9	6	8
Secchi Depth	77	100	120	170	150	120	NA	NA	NA
1 % Light	2.3	3.0	3.5	4.0	4.0	3.5	NA	NA	NA
pH	8.1	8.0	8.2	8.2	8.5	8.0	7.5	7.7	8.0
Temperature	14	14	13	15	14	14	15	15	12
Dissolved Oxygen	8.0	8.6	8.6	9.1	9.0	9.8	9.0	8.8	10.0
Conductivity	250	250	250	240	245	250	370	250	220

* = no data